Bidhan Chandra Patra
Pravat Kumar Shit
Gouri Sankar Bhunia
Manojit Bhattacharya *Editors*

River Health and Ecology in South Asia

Pollution, Restoration, and Conservation

Springer

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Chapter 1
River Health and Ecology: Perspective View and Approach

Bidhan Chandra Patra, Pravat Kumar Shit, Gouri Sankar Bhunia, and Manojit Bhattacharya

Abstract In the twenty-first century, population explosion and anthropogenic pressure are vital problems on the global scale and have reflected crucial effects on our environment as well as river ecosystem and health. Anthropogenic activities have broken the original ecological balances and affected the natural structure and functions of the innate river ecosystem and, consequently, gradually degraded the river health system, functions, and ecosystem services. Nowadays, a burning issue is how to restore and scope the damaged river ecosystem back to a healthy status. Important effective ecological restoration measures are urgently needed to be implemented in our society. This chapter summarises the various aspects of river health, human interferences, remediation, and pollution management strategies based on the present approach and provides a research assessment for future research and advancement.

Keywords River health · Human interferences · Ecological restoration · Remediation and pollution management strategies

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1 Introduction

A river or watercourse is known as the channel, the riparian zones (such as floodplain wetlands), and the estuary. River conservation has made the preservation and protection of 'healthy' river habitats a priority. Rivers are valued by most people as reservoirs of safe water for drinking and bathing, for industrial and agricultural applications, as pollutant conduits, for leisure and scenic enjoyment, for fisheries, and for a variety of other purposes. As a result, since certain water quality issues are intimately linked to the relationship of rivers and their catchments, riverbank, and wetlands, a regional scale solution is often the most suitable. Since rivers are complex physical, chemical, and biological systems, these operations must be multifunctional to make a proper chance of success (Norris and Thoms 1999). River health assessments should be precise, prompt (warning of worsening rather than waiting until the patient is dying), fast (to allow for quick response), and affordable. River management examines features of various biological, chemical, engineering, and geomorphological strategies. There has recently been a movement toward using biological approaches to measure river health (Boulton 1999). This is because biota impacts are normally the last stage of environmental destruction and river contamination. As a result, river security standards have changed their emphasis from mostly physical and chemical interventions (on the premise that if they were met, an optimal river state would be achieved) to more biological measures.

However, the term 'river health' is fundamentally contradictory since it incorporates natural differences in shape and function that occur across all river systems (Nandi et al. 2016). The real condition of the river does not constitute the river's health until it is tested against a set of well-defined criteria. Many people have embraced the word 'river health,' which is also used as similar to human health and provides a general sense of understanding as a result of the current focus on biota and habitats. The River Health Assessment (RHA) idea arose from a need to assess the river's health. For a long time, these experiments were solely concerned with water quality, which involved physicochemical properties. RHA uses a variety of metrics to assess river health, including catchment, floodplain, course of the river, water flow, water quality, and biological health. Both river ecologists and managers must be able to characterise river quality in some way, recognise main indicators, and distinguish between natural variation and criteria that indicate human impacts on river health (Thoms et al. 1999). Karr (1999) examines the history of the idea of stream health and, like Meyer (1997), provides a compelling case for including ecological honesty (the preservation of environmental erection and purpose) and anthropological ethics (what community values in the ecosystem) in the meaning of river health. Unlike conceptions of sustainable habitats based solely on biological parameters, river health judgments must take into account human interests, uses, and services resulting from the structure. River ecologists may make a significant contribution, but they must accurately convey the knowledge and assist in the direction of the study effort if their results are to be used to preserve and assimilate river health.

River health is characterized as 'the capacity of the water habitats' performance and retention of key ecosystem functions and a population of creatures with a species distribution, multiplicity, and functional organization that is similar to that of uninterrupted ecosystems in the region as possible.' A river that is ecologically sound would have flow regimes, aquatic eminence, and channel features such as the following:

- The natural functions of the environment are preserved.
- The major characteristics of natural habitats are portrayed and are preserved over time.
- Natural riparian forest ecosystems can be found for most of the river's reach.
- The river allows aquatic fish and other fauna to float up and down.
- Linkages between the river and the floodplain, as well as related wetlands, are capable of maintaining ecological cycles.
- Natural connections to the sea or to the terminal lakes are preserved.

2 River Health: Issues and Challenges

Environmental flow (E-flow) has gained in popularity in recent years, with a growing recognition of the value of maintaining any water in a river to preserve the ecosystem's well-being (Tharme and King 1998; Wei et al. 2012). Rivers are diverse habitats that have been seriously harmed, if not completely ruined, as a result of overexploitation and population growth. Water habitats have been degraded and significantly changed as a result of the close interaction between waterways and human populations as well as socio-economic trends. Many research on river health assessments have been carried out to avoid further degradation of the river ecology (Boulton 1999; Pont et al. 2007; Magagula et al. 2010; Patra et al. 2017).

The River Health Assessment (RHA) has piqued the interest of inventors and policymakers all over the biosphere, resulting in the launch of many RHA ventures around the world. Australia looked for quick biological evaluations to take actions, strategies, and legislative frameworks as a way of evaluating: (a) levels of achievement towards defined and clear quality ideas; (b) advancement towards meeting clear goals for enhanced environmental quality; (c) possible consequences to aquatic ecosystems from anthropogenic activities; and (d) the climate changes (Australia-Wide Assessment of River Health 2001). However, the goals of a countrywide evaluation of the status of wade able streams and rivers in the United States were to (1) on a regional level, assess the current state of main metrics on the status of the Nation's streams and tributaries with known output assurance and (2) assess the status of wade able streams and rivers in the United States and search for correlations between chosen measures of natural and anthropogenic impacts (USEPA 2004). The seven Chinese river commissions researched to streamline a river health requirement that was appropriate for their respective river basins, with the aim of supporting the goals of the Chinese government by improving water quality and river health in the Gui, Pearl, Lio, and Yellow river basins (Gippel et al. 2011).

In the case of India, rivers are the primary source of life for the country's vast population. In India, the rivers are experiencing an extreme crisis. The volume of polluted rivers in the world has more than doubled throughout the last five years, according to a 2013 estimate by the Central Pollution Control Board (CPCB). This is mostly attributed to unprocessed waste and toxic sewages being dumped into waterways, placing a burden on the riverine environment and rendering it unsafe. Excluding the Yamuna Jiye Abhiyan's evaluation of the river's health in Jalalpur (Allahabad), the RHA's effort has primarily centred on the water quality catalogue. About the fact that the Ganga forms India's prime river catchment, there is a considerable deficiency of knowledge about the river's state. Just a few aspects of the RHA protocol were included in the Ganga Action Plan I (GAP I), which was implemented in 1986 with the objective of cleaning the river. Contamination from crop waste, human excretions, livestock swimming, and the drainage of unburned and half-burned corpses into the river was among them as were the restoration of the river's biotic (plant, animal, and ecosystem) diversity and soft-shelled clam regeneration (https://nmcg.nic.in/gangaactionplan1.aspx). The second phase (GAP-II) of the project began in 1993, with construction on four tributaries of the Yamuna, Gomti, Damodar, and Mahanadi rivers. In 2011, the Indian government, with the support of the World Bank, initiated another clean-up project, the National Ganga River Basin Project. According to the program's findings, stakeholder interest in pollution management programs is critical in any attempt to clean, rebuild, and recreate the River Ganges catchment. A new assessment of the Chenab River's water quality during the recession period in India measured the water quality index (WQI) for three different water uses (irrigation, drinking, and aquatic life) and concluded that the river Chenab's overall WQI rating was weak and unhealthy for drinking, while the quality of water has improved for both agriculture and aquatic species (Bhatti and Latif 2011).

3 Key Aims of the Book

Rivers play numerous imperative roles in the well-being of people and agricultural production as well as river ecosystem; however, severe human activities have broken down the original ecological balances and affected the natural structure and functions of the innate river ecosystem. To restore and scope the damaged river ecosystem back to a healthy status, important effective ecological restoration measures urgently need to be implemented in our society. The main problems that the spoiled rivers face are the locally altered hydrological processes affected by the construction of hydraulic facilities, the deterioration in water qualities resulted from unregulated pollution emissions, and many other ways. In this book, we tried to address and follow-up on the main ecological restoration techniques of the rivers affected by undesirable changes or pollution that were reviewed and summarised, respectively. Simultaneously, numerous manmade activities also reflect on the direct damage of natural environmental aspects. In addition, three key methods, that is physical, chemical, and eco-biological techniques are purposefully introduced in detail aims to a reduction in riverine water pollution and remediation strategies through case studies.

4 Individual Chapters

There are 18 chapters in this book to address anthologies actives on river health and ecology. Chapter 2 deals with bio-indicator to decipher the health of the water of South Asian Rivers.

The spatial and temporal variation in physiochemical properties of river water of Kangsabati River has been described in Chap. 3. The consequence of human activities on river ecology is illustrated.

The chapter concluded androgenic tresses like sand mining, expansion of agricultural field toward river and its encroachment, excessive use of chemical fertilizer and pesticides in crop land adjacent to river, construction of a temporary dam on the riverbed to store the water arresting river flow for the purpose of irrigation, unscientific construction of bridge and embankment, the introduction of exotic fish and plant species, and so on. Chapter 4 deals with this aspect to estimate the role of modern biotechnology in the era of river water pollution and use and implement advanced scientific technologies, such as microbiology, molecular biology, biotechnology, bioinformatics, Metagenomics, and other omics platforms to observe, analyse, and manage the information obtained from aquatic ecosystems to develop superior biotechnological methods to maintain the quality of freshwater. Chapter 5 deals with the assessment of chitinolytic bacteria isolated from zooplankton of the freshwater ecosystem. The results obtained have unearthed some new and interesting research information pertaining to the trophic interactions through the exchange of biochemical entities in an aquatic ecosystem, the baseline knowledge of which are expected to be utilized in the ecosystem management from the point of view productivity and also of maintaining of the ecosystem health. Chapter 6 discusses microplastics in freshwater riverine systems and provides a scientific overview of the present situation. Chapter 7 deals with the evaluation of the ecosystem health of River Mahanadi and its seven tributaries at the nodes by analysing the physical habitats, chemical characteristics of water, and biological attributes of fishes. Chapter 8 illustrates the role of remote sensing and GIS techniques on how to prepare the surface water and water quality delineation map. This chapter also emphasizes on the various spectral signature on the Landsat TM and OLI image and extracts all the inventory indices of water bodies such as NDWI, NDTI, and so on. Chapter 9 deals with this aspect with anthropogenic impacts on hydro-morphodynamic behaviour in the middle and lower courses of river Subarnarekha, India. The chapter also highlights that the unscientific anthropogenic activities directly or indirectly create a harsh impact on the natural behaviour of the river as well as on the riparian dwellers.

Consequently, the channel margin land erosion and bank failure are accelerated at the position of mining sites and in the downstream section and opposite bank of the embankment structures due to change in hydrodynamic behaviour of the river. Chapter 10 discusses the toxicology of riverine pollution caused by organic and inorganic contaminants and pollutants-induced perturbations of the riverine ecosystem at a cellular and molecular level.

Chapter 11 evaluates habitat quality in quarried reach of the alluvial river. The chapter also highlights the status of channel and ecological attributes in channel bed, riparian, and bank sites; identifies the correlation between channel attributes and ecological attributes on growing hydro-ecological succession and finds out the threshold value of a more effective variable for further reaching the resilience state to manage the river restoration at patch scale. In Chap. 12 written by a group of researchers, the chapter discusses the hydro-chemical and microbial indicators for water quality assessment in an industrial catchment of river Damodar, India. Chapter 13 examines the heavy metal concentration of fish species in the peninsular river of Subarnarekha, India. The heavy metal concentration of Cd, Cu, Pb, Ni, Mn, Zn, and Cr were estimated in three fish species (*Puntius ticto, Glossogobius guiris, and Labeo bata*). Chapter 14 deals with the Spatio-temporal variation in water pollution of the Churni River based on the water quality index and aquatic toxicity index. Chapter 15 demonstrates the current and future geospatial strategies that can be used to better understand the river corridor and ecology straggles. Chapter 16 presents an endeavour to define the microplastic pollution in freshwater systems and a piece of state-of-the-art information about microplastics, risks associated, and possible remediation techniques. Chapter 17 deals with anthropogenic activities on riverbed sand and gravel mining and consequences of human well-being of Raidak-II River in Eastern Dooars, India. The last chapter of the book describes the impact of human activities on the freshwater environment and its consequences through field observation as well as google earth images analysis. This chapter also emphasizes the various remediation and pollution management strategies based on the present approach and provides a research assessment for future research and advancement. All chapters cover extensively the literature and present new results and management ideas through ground observation, laboratory experiment, and field knowledge for future work.

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Chapter 2
Arthropods: An Important Bio-Indicator to Decipher the Health of the Water of South Asian Rivers

Puja Karmakar, Suchismita Pal, and Monalisa Mishra

Abstract The river is an open ecosystem that can be easily influenced by human activities. River pollution in South Asian countries is increasing day by day. The status of the water pollution of the river is measured using different indicators. Biomonitoring is one of the most common and inexpensive methods to detect the water quality of the river. Aquatic arthropods are often used as bioindicators in water bodies. Any change in the population, diversity, and life cycle throughout the year can indicate the water pollution level. Among arthropods, class Insecta and crustacea are used as environmental bioindicators. The insect family, which can be included as bioindicators, includes Ephemeroptera, Plecoptera, Coleoptera, Diptera, and Odonata. Similarly, among Crustacea, Cladocerans, Ostracods, Decapoda, and Amphipoda are commonly used as bioindicators. Any altered population and species number in these families is regarded as an indicator of pollution. The current chapter summarises all the approaches used to detect the health of South Asian Rivers using arthropods as bioindicators.

Keywords Water pollution · Biomonitoring · Bioindicator · Insecta · Crustacea · South Asian rivers

1 Introduction

Human or anthropogenic activity can impact the terrestrial as well as the aquatic environment. As a result, structural and functional changes are seen in the aquatic ecosystem. The structural changes include irrigation, damming, and an increase in sedimentation (Dudgeon et al. 2006; Thorne and Williams 1997). The structural modification affects the functional modification in the habitat. The functional

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modification further occurs when there is an improper discharge of wastewater and industrial discharge into the water. This functional modification ultimately influences the nutrient status, light penetration ability, and depth of the water bodies (López-López and Sedeño-Díaz 2015). As a result, the freshwater ecosystem (e.g. river) is the most endangered in the world. The most affected rivers are found in developing countries due to high population, urbanisation, and emerging industrial processes (Dudgeon et al. 2006; Mustow 2002). It is challenging for water animals to live in a polluted ecosystem. Either they undergo modification to cope up with the environmental change or they perish with time (Livingston et al. 1994). Thus, water pollution is one of the major causes of species extinction in an aquatic environment (Livingston et al. 1994).

Among all the animal species of the world, 10% of animals live in the freshwater ecosystem (Dudgeon et al. 2006). Thus, the protection of the waterbodies is an alternative to conserve them (Balian et al. 2007; López-López and Sedeño-Díaz 2015). In Asia, mainly in South Asia, many great rivers are known to be the cultural and economic backbone of the continent. As a result, all around South Asia, the rivers have acute pollution problems (Subramanian and Sivaramakrishnan 2007) (Ramachandra and Solanki 2007). The factors responsible for the alteration in running waters include ecological destruction, physical and chemical alteration of the water, and direct presence or absence of species (Archan et al. 2015). The Asian freshwater ecosystems possess an exceptionally rich ecosystem (Azam et al. 2015; Barman and Gupta 2015; Ganguly et al. 2018; Lonkar and Kedar 2013; Sharma et al. 2013; Shukla et al. 2016). In this context, assessment of the health of river water is an alternative to protect the aquatic fauna present in those rivers. The physical and chemical parameters of water bodies are measured to detect the health of waterbodies (Li et al. 2010; Malmqvist and Rundle 2002; Resh and Rosenberg 1993). These two are widely accepted methods in various countries to monitor the quality of river water (Couceiro et al. 2012; Gerhardt 1996; Li et al. 2010; López-López and Sedeño-Díaz 2015; Metcalfe 1989; Ortiz-Ordonez et al. 2011; Reece and Richardson 1999). However, the hydrology of river water is not stable (Li et al. 2010; Soininen and Könönen 2004) which makes it difficult to detect the actual state of the pollution. To overcome this problem, the supplementary method has been developed (Li et al. 2010) which is less expensive and easy to detect the level of water pollution (Li et al. 2010). This method uses the abundance of the aquatic organisms at different water levels (Biggs 1989; Cairns and Pratt 1993) which are termed bioindicators. The current chapter summarises the use of arthropod as a bioindicator (López-López and Sedeño-Díaz 2015) or a biomonitoring agent (Subramanian and Sivaramakrishnan 2007) to detect the health of South Asian rivers.

2 Monitoring Pollution Using Bioindicator

The concept of bioindicator traces back to Aristotle who observed the reactions of the freshwater fish into the seawater (Markert et al. 1995). Later, there are many concepts regarding the 'biological indicators' of pollution. In the early 1900s, Kolkwitz and Marsson introduced the concept of bioindicators. As stated by Markert et al. a bioindicator is 'an organism (or part of an organism or a community of organisms) that contains information on the quality of the environment (or a part of the environment)' (Markert et al. 1997; Markert et al. 1999; Markert et al. 2003).

2.1 *Various Characteristics that Make an Organism a Bioindicator*

An organism must satisfy the following criteria to be used as a bioindicator

(a) The species should be easily identified without specialists.
(b) The species should have wide distribution.
(c) The species should confine to a particular locality.
(d) The characteristics of the species should have well-known ecological characteristics.
(e) The species should be available in a good number.
(f) It can be easily used for laboratory experiments.
(g) It should be sensitive to any kind of environmental stress.
(h) Easy for quantification and standardisation. (Füreder and Reynolds 2003; Hilty and Merenlender 2000; Resh and Rosenberg 1993).

2.1.1 How to Select a Bioindicator?

To measure the level of pollution in the river, a huge amount of money, effort, expertise, and time is required. To overcome this, conservationist uses various fauna to monitor the health of the water bodies. The richness of species in an ecosystem is considered a major bioindicator of the waterbodies (Stevenson et al. 1996). Some species are sensitive to specific environmental stressors that influence their growth and population size**.** Such species are known as indicator species (Li et al. 2010; Rinderhagen et al. 2000; Rosenberg 1998). The presence of the number of indicator species reflects the estimation of the degree of environmental impact and its potentially harmful impact on the living organisms. Various types of indicators and their functions are summarised in figure (Fig. 2.1).

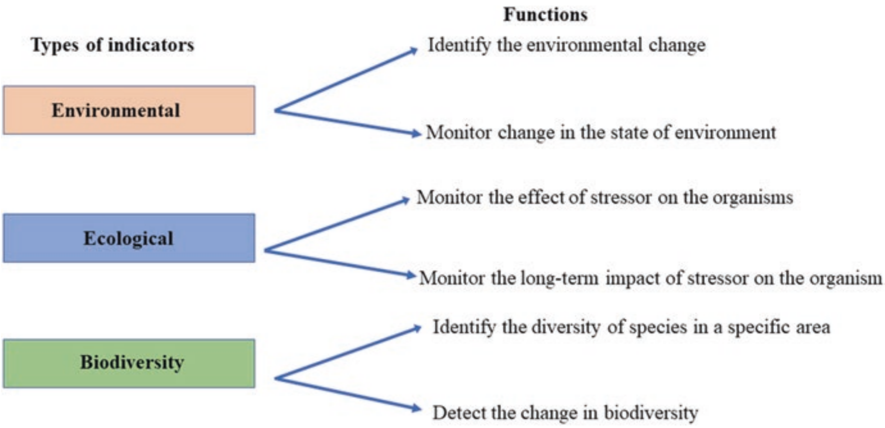

Fig. 2.1 Types of bioindicators and their functions in the ecosystem

2.2 *Biomonitoring and Bioindicators*

- Biomonitoring is a method in which specific organisms are used to measure the water quality (chemical contents of the water) (Markert et al. 2003). The quantification of the organism reflects the factors related to the water pollution of the river. Thus, any change in its number, life cycle, and distribution reflects the quality of water pollution (Subramanian and Sivaramakrishnan 2007). The biomonitoring can be done at sub-organismal, organismal, population, community, and ecosystem levels (Li et al. 2010). During the process of biomonitoring, either the structure or the mechanism gets affected at various levels of organisation (Adams and Greeley 2000; Bartell 2006). The effect seen at the organism level is in terms of chemical, physical, and biological levels (Cota et al. 2002). The biological responses seen towards the contaminants follow sequentially in a developmental time window. First, the contaminants present in the ecosystem contact with the aquatic organism orally, through skin and gills, and so on. Once the contaminant entered the body, it affects the organism at molecular, physiological, and morphological levels. All these changes finally alter the behaviour of the animal (Fig. 2.2.). At the sub-organismal level, the damage to the organism can be measured by early warning methods thus used as bioindicators. With time, this damage is seen at various levels. The long-term effect of damage can be seen at the population, community, and ecosystem levels. All these disturbances affect the nutrient cycle, decrease productivity, disturbed the food chain, and altered inter and intra specific interactions among the organisms (Sedeño-Díaz and López-López 2013) (Figs. 2.2 and 2.3).

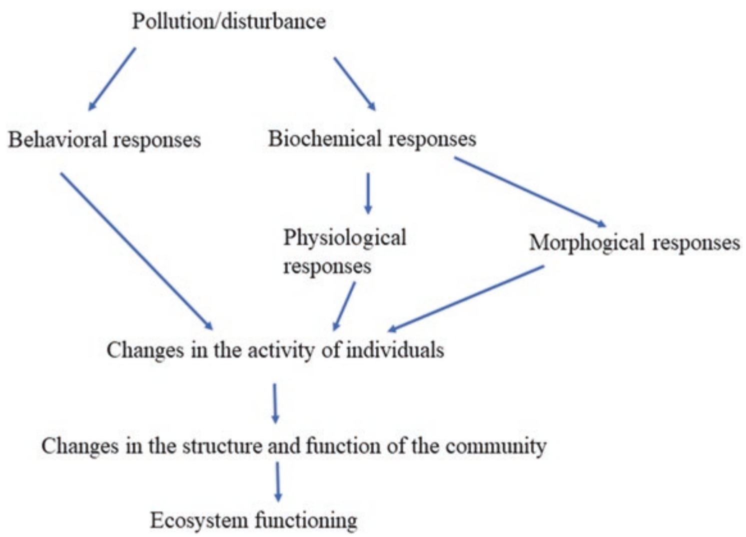

Fig. 2.2 Sequential development of biological response at different organisation levels (Redrwan from Adams and Greeley 2000; López-López and Sedeno-Diaz 2015)

2.3 Various Advantages of Biomonitoring

Biomonitoring is a useful method to detect the health of river water for the following reasons (Central Pollution Control Board 1995),

1. Chemical detection is a time-consuming process, and it needs many chemicals.
2. Some of the chemicals are not adequately present in the water. Thus, the finding of such chemicals in water can be monitored by pollution-sensitive instruments.
3. Chemicals toxicants behave differently when it is in mixture form. Thus, the actual nature of the toxicants cannot be detected.
4. The chemical property of the toxicants changes with the environment. But biologically, such changes can take place for a long period in terms of effect that can be easily measured (CPCB & Intra-portal 1995; Li et al. 2010).

2.3.1 Biomonitoring Approaches Based on Hopkin (1993):

The presence or absence of taxa in a specific region should be monitored. The concentration of the pollutants should be used as an indicator or sediment.

- The concentration of the pollutants impacts the organism's biotic and abiotic factors. It should be able to detect different strains that are resistant to pollutants (Hopkin et al. 1993).
- Many organisms are used as bioindicators to assess the effects of pollution on the freshwater ecosystems—algae, protozoa, periphyton, fishes, micro invertebrates,

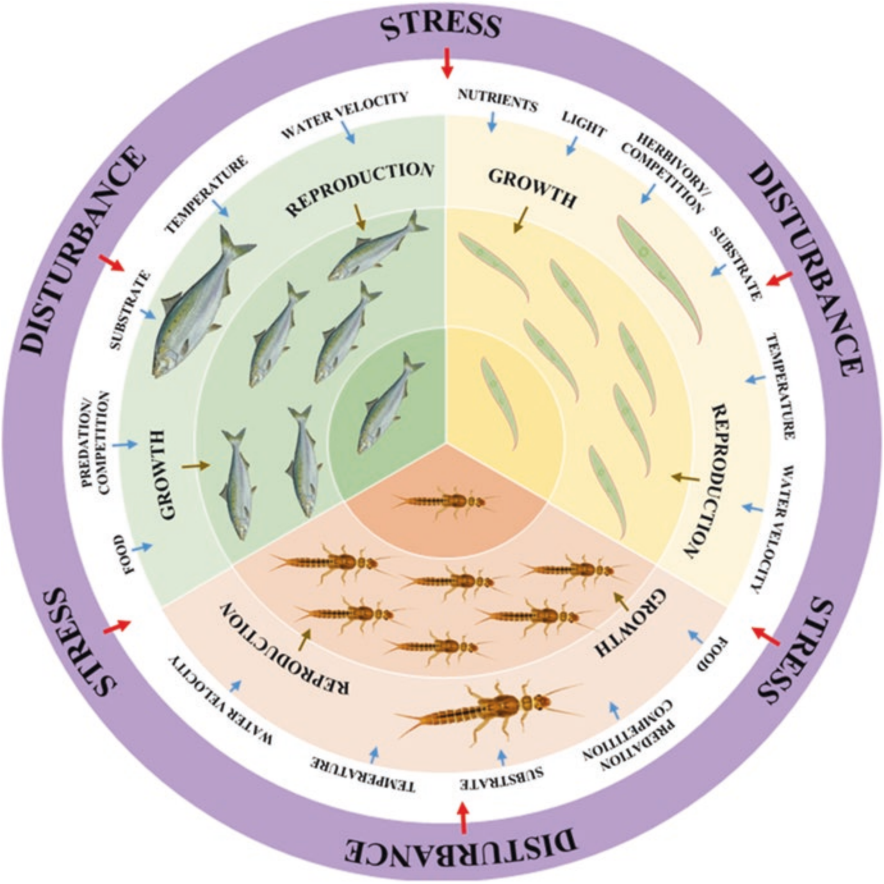

Fig. 2.3 Concept of the biomonitoring**.** Representation of a freshwater ecosystem. Physical and chemical factors of the water control the aquatic ecosystem. Change in the physicochemical status of the water regulates the diversity, abundance of the fauna in the ecosystem. In stress conditions, abiotic and biotic factors of the ecosystem (e.g. light, substrate, temperature, water velocity, competition, and availability of food) change the environmental condition, and as a response, it affects the normal growth, reproduction ability of the organism. The representative organisms are worms, insects, and fishes Redrawn from Bioindicator. (2017, August 18). Retrieved September 22, 2021, from Alchetron.com website: https://alchetron.com/Bioindicator

and macroinvertebrates (Birk and Hering 2009; Buffagni et al. 2004; Carlisle et al. 2008; Coste et al. 2009; Fausch et al. 1990; Joy and Death 2002; Oberdorff et al. 2002; Pont et al. 2006; Resh and Rosenberg 1993; Scuri et al. 2006; Soininen and Könönen 2004; Statzner et al. 2001; Torrisi et al. 2010; Vis et al. 1998; Whitton and Rott 1996; Whitton and Kelly 1995).

- Many studies revealed two concepts of bioindicators: the first population bioindicator was developed in 1942 and the second biodiversity bioindicator in 1980 (Noss 1990). It makes a complicated concept of biodiversity or ecosystem into

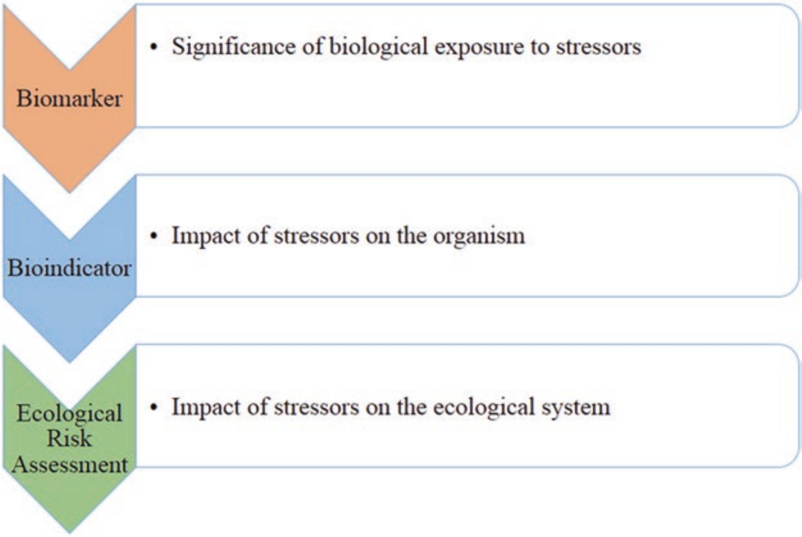

Fig. 2.4 Design of a bioindicator (modified by Jain et al. 2010)

easy, manageable, and specific quantifiable indicators (Noss 1990). Bioindicators are classified into two types (Fig. 2.4).

1. *Accumulation bioindicator*: These types of bioindicators can store the pollutants into their body without any change in their metabolism (Jain et al. 2010).
2. *Response bioindicator*: These types of bioindicators are very sensitive and responsive to any subtle changes in the surroundings (Jain et al. 2010). Types of responses are three types:

- Ecological changes: These changes involved species diversity, population density, and key species (Jain et al. 2010).
- Behavioural changes: These are related to the changes in feeding activities, the presence of bacteria, and mobility (Jain et al. 2010).
- Physiological changes: These are related to heavy metal accumulation, microbial activity, CO_2, and Biochemical Oxygen Demand (BOD) levels in the ecosystem (Jain et al. 2010) (Fig. 2.5).

In the river ecosystem, benthic invertebrates are the most stable, long-lived, and very delicate communities. Because of these qualities, they are used as the most trusted and informative bioindicators of water. Their presence in the habitat is related to the physical and chemical characteristics of the substrate (Hynes 1960). The specific response of benthic macroinvertebrates is seen towards the alteration of the water quality, the integrated impact of food, and predation. Seasonal changes are also known to influence the diversity of benthic invertebrates (Emere and Nasiru 2009; Groenendijk et al. 1998). Studies on aquatic arthropods have frequently monitored the relationship of species, habitat, and the pollution of the habitat. Some

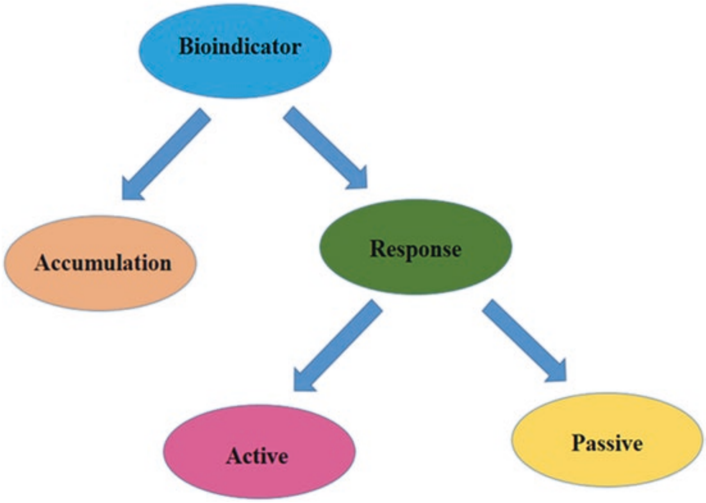

Fig. 2.5 Categories of bioindicators (modified by Jain et al. 2010)

species need certain resources for their growth, and once all resources are available in habitat, the species is tolerant of the factors present in the habitat. Thus, that species is considered to belong to that habitat or ecosystem (Rosenberg 1998).

3 Various Aquatic Organisms used as Bioindicators

The physical and chemical characteristics of the rivers are an important way of identification of the nature of water for an aquatic ecosystem. The physicochemical characteristics of water, food resources, and substrate resources are important in controlling the dispersion of the organisms, that is Arthropods (Patang et al. 2018; Subramanian and Sivaramakrishnan 2007). Arthropods are aquatic invertebrates that are sensitive to the anthropogenic changes in the river ecosystem. Thus, any disturbance in the anthropogenic parameter may be the reason for the inhibition of the taxa. Water pollution is a serious problem that influences the community of organisms of the streams (Johnson and Stinchcombe 2007). Conditioning factors such as environmental heterogeneity, applied management system, seasonality, water characteristics, types of vegetation cover, availability of litter, anthropic activities, temperature, pH, and conductivity commonly influence the community of the organisms present within the river (Johnson and Stinchcombe 2007; Patang et al. 2018). Many organisms that are sensitive to these changes are used to detect the nature of the water in the rivers (Archan et al. 2015; Patang et al. 2018; Shukla et al. 2016; Subramanian and Sivaramakrishnan 2007). The organisms that are used for biomonitoring include Fish, Algae, Mussels, Bacteria, and macroinvertebrates. Macroinvertebrates include a different class of organisms that live in the aquatic ecosystem and have a size of more than 250 m. Thus, they can easily be monitored using a net. Macroinvertebrates spend some stage of life in water, and they are mainly benthic. They can be mainly classified as (1) non-arthropod macroinvertebrates and (2) arthropod macroinvertebrates. The non-arthropod macroarthropod includes: Platyhemminthes (e.g. flatworm), annelids (e.g. earthworm and Leech), molluscs (snails and bivalves), and mites. Among macroarthropods, the class Insecta, Crustacea, are the most dominating groups of organisms in the freshwater aquatic ecosystem. A total of 1,302,809 species of arthropods have been reported in the aquatic ecosystem (Chakravarthy et al. 2016; Zhang 2013) out of which 1,070,781 species of insects are associated with the aquatic ecosystem, which accounts for 80% of the Arthropod species (Chakravarthy et al. 2016; Zhang 2013). Aquatic insects are diverse and present in different aquatic environments. A total of 45,000 different species of insects are known to be found in the different freshwater ecosystems (Gogoi and Guha 2000) out of which many of them spend their part of the life cycle in water. Approximately less than 3% of all insect species have early developmental aquatic stages in freshwater ecosystems. The developmental stages of insects comprise about 95% of the total biomass of macro-invertebrates of the freshwater ecosystem. Thus, they play a major ecological importance in maintaining the freshwater ecosystem (Gogoi and Guha 2000). Some groups of insects are very sensitive to the changes in their habitats. Aquatic insects have a fixed growth, developmental cycle, and typical morphological character (Shrivastava et al. 2018; Subramanian and Sivaramakrishnan 2007). These features of aquatic insects make them an indicator of pollution. Many studies reported that the health of the river can be assessed from the diversity and abundance of the insects in their habitats (Barman and Gupta 2015; Gogoi and Guha 2000; Shrivastava et al. 2018; Subramanian and Sivaramakrishnan 2007). Insects respond quickly to the changes in their habitats, and these are good bioindicators; therefore, knowing the diversity of species and their degree of interaction with the environment analyse the impacts of anthropogenic modifications in the ecosystems (Paoletti 1999) (Table 2.1). For example, stonefly (Plecoptera) larvae are very responsive to alterations in water quality (Kimberling et al. 2001; Paoletti 1999). On a scale of 0 to 10, 10 is known to be the tolerant species, and the rank of stonefly larvae was reported between 0 and 2 depending on the specific species (Azrina et al. 2006).

4 Various Arthropod Species Used as a Bioindicator

Arthropods are the most successful animal on this earth in terms of numbers as well as species. It is difficult to find a place that is free from arthropods. Rivers are also enriched with various Arthropods. Thus, they are the best indicators to detect the level of water pollution (Imoobe and Ohiozebau 2010). Growth and the developmental cycle of many Arthropods are closely associated with the physical properties of the water bodies (Callanan et al. 2008). Thus, any alteration in the water body will affect the survivability of the arthropods present in the aquatic ecosystem.

Table 2.1 List of aquatic insects as bioindicators used in South Asian countries

Order	Family	Taxa
Hemiptera (aquatic bugs)	Gerridae	*Ptilomera assamensis* Barman and Gupta (2015), Gupta and Chaturvedi (2008)
		Pleciobates expositus Barman and Gupta (2015)
		Metrocoris nigrofascioides Barman and Gupta (2015)
		Ovatametragualeguay Barman and Gupta (2015)
	Aradidae	*Notapictinus Aurivilli* Barman and Gupta (2015)
	Mesoveliidae	*Mesovelia vittigera* Barman and Gupta (2015)
	Veliidae	*Rhagovelia sumatrensis* Barman and Gupta (2015)
Coleoptera (aquatic beetles)	Gyrinidae.	*Orectogyrus sp.* Barman and Gupta (2015)
		Dineutus sp. Barman and Gupta (2015)
	Curuculionoidae	*Bagous affinis* Barman and Gupta (2015)
	Noteridae	*Hydrocanthus oblongus* Barman and Gupta (2015)
Trichoptera (caddies fly)	Hydroptilidae	*Hydroptila sp.* Barman and Gupta (2015)
	Hydropsychedae	*Hydropsyche bidens* Barman and Gupta (2015)
Ephemeroptera (mayfly)	Baetidae	*Labeobaetis sp.* Barman and Gupta (2015)
		Offadens sp. Barman and Gupta (2015)
	Heptageniidae	*Stenonema sp.* Barman and Gupta (2015)
	Caenidae	*Heptagenia sp.* Barman and Gupta (2015)
Odonata (dragonfly and Damse fly)	Gomphidae	*Erpetogomphus sp.* Barman and Gupta (2015)
	Chlorocyphidae	*Libellago lineata indica* Shukla et al. (2016)
	Libellulidae	*Brachythemis contaminata* Shukla et al. (2016)
	Platycnemididae	*Copera marginipes* Shukla et al. (2016)
	Lestidae	*Lestes umbrinus* Shukla et al. (2016)
	Aeshnidae	*Anax guttatus* Shukla et al. (2016)
		Gynacantha bayadera Shukla et al. (2016)
		Hemianax eohippiger Shukla et al. (2016)
	Coenagrionidae	*Agriocnemis femina* Shukla et al. (2016)
		Agriocnemis pygmaea Shukla et al. (2016)
		Agriocnemis pieris Shukla et al. (2016)
		Ceriagrioncoromandelianum Shukla et al. (2016)
		Disparoneuraquadrimaculata Shukla et al. (2016)
		Enallagmaparvum Shukla et al. (2016)
		Ischmura aurora Shukla et al. (2016)
		Ischmurasenegalensis Shukla et al. (2016)
		Pseudagrionsepencei Shukla et al. (2016)
		Rhodiscnuranursei Shukla et al. (2016)
		Pseudagrion decorum Shukla et al. (2016)
		Pseudagrionrbriceps Shukla et al. (2016)
Collembola	Entomobryidae	*Entomobryanivalis* Archan et al. (2015)
		E.sp. Archan et al. (2015)
Plecoptera (stone fly)	Chloroperlidae	*Xanthoper laacuta* Archan et al. (2015), Babu et al. (2019)

(continued)

Table 2.1 (continued)

Order	Family	Taxa
Orthoptera (grasshoppers and crickets)	Tetrigidae	*Agrion Euparatettixtenuis* (Archan et al. (2015), Shrivastava et al. (2018), Subramanian and Sivaramakrishnan (2007)
Diptera (Fly)	Dixidae	*Nothodixa sp.* Archan et al. (2015), Subramanian and Sivaramakrishnan (2007)

Henceforth, they can be widely used as bioindicators of water pollution and the health of rivers. We are summarising the most popular bioindicator of insects and Crustaceans order subsequently.

4.1 Class: Insecta

Ephemeroptera (May Fly) A few species of the order Ephemeroptera are very sensitive towards pollution (Archan et al. 2015). Family members of Ephemeroptera like Baetidae, Canidae, Ephemerellidae, Leptophlebiidae, Potamanthidae, Trichorythidae, and Oligoneuridae are commonly found in the rivers of some South Asian countries (Subramanian and Sivaramakrishnan 2007). The species of Ephemeroptera is not tolerant of organic enrichment. The family Baetidae is an exception, which can tolerate the high organic substances in the water. In rivers of India, the percentage of Ephemeroptera varies with seasons. In monsoon, the percentage is higher than in the summer and winter. More amount of Ephemeroptera in the rainy season indicates the presence of less pollution in the water during the monsoon (Archan et al. 2015).

Trichoptera (Caddis Fly) They are the largest group of aquatic insects and play an important role in the aquatic ecosystem (Subramanian and Sivaramakrishnan 2007). They process organic materials. The popular bioindicator families of Trichoptera are Hydroptilidae, Hydropsychedae, Calamoceratidae, Glossomatidae, and Helicopsychidae, Lepidostomatidae, Limnephilidae, Philopotamidae, Polycentropodidae, Rhyacophilidae, and Stenopsychidae (Archan et al. 2015; Subramanian and Sivaramakrishnan 2007). The study on the river Kshipra of India revealed that the seasonal percentage dominance of Trichoptera was 6.49%, 6.35%, and 0.62 in monsoon, winter, and summer, respectively. The Trichoptera group of insects is sensitive to pollution but can tolerate pollution if enough nutrients and oxygen are present in the water (Archan et al. 2015).

Plecoptera (Stone Fly) The order Plecoptera is known to be the less abundant group of insects in the river water (Archan et al. 2015). They are mostly present in the cold and moving water with enough oxygen. The order Plecopterais is represented by family Capiniidae, Chloroperlidae, Leuctridae, Nemouridae, Peltoperilidae, Perilidae, Perlodidae, and Taeniopterygidae (Subramanian and

Sivaramakrishnan 2007). The number increases during the monsoon. More amount of the species during monsoon indicates the good quality of water (Archan et al. 2015; Rao et al. 1987).

Coleoptera (Beetles) Coleoptera is the most popular order of the Class Insecta. They are commonly known as beetles. Water beetles are the most diverse groups of insects. They are known as an ideal bioindicator for their high sensitivity to polluted water (Archan et al. 2015). Gyrinidae, Curuculionoidae, Noteridae, Hydrophilidae, Psephenidae, Dytiscidae, Haliplidae, Elmidae, and Staphylinidae are some important bioindicator groups of insects (Subramanian and Sivaramakrishnan 2007). Their presence is more in river water during monsoon. A higher amount of Coleopteran species during rain further indicates the good quality of water (Archan et al. 2015).

Hemiptera (Water Bugs) They are commonly known as water bugs. This is a large order of the class Insecta. The aquatic bugs can live both in running and in standing water. Several species of Heteroptera are very sensitive to polluted water. The order of those species is represented by the family Notonectidae, Gerridae, Belostomatidae, Corixidae, and Nepidae (Archan et al. 2015; Subramanian and Sivaramakrishnan 2007). The complete absence of Hemiptera during summer is reported in many studies. The absence of Heteropatera during summer indicates the level of water pollution in summer (Archan et al. 2015).

Odonata (Dragonflies) Odonates are excellent bioindicators of aquatic habitats. They are widely used by conservationists as ideal surrogate taxa. Odonates inhabit both terrestrial and aquatic habitats during their life cycle. The presence of these insects in water marked organic pollution in the water. Order Odonata is represented by family Gomphidae, Chlorocyphidae, Euphaediae, Libellulidae, Protoneuridae, Coenagriidae, and Corduliidae (Archan et al. 2015; Subramanian and Sivaramakrishnan 2007). The complete absence of Odonates species in summer indicate the high pollution of the river water of India (Lonkar and Kedar 2013; Shukla et al. 2016). Among Odonata Dragonfly and damesfy nymphs were used as bioindicator in Ambazari Lake, Nagpur, India (Lonkar et al. 2015).

Diptera Diptera lives in a wide range of habitats. They are the most popular bioindicator of polluted water. Diptera represents the largest family of aquatic insects. Order Diptera is represented by family Chironomidae, Culicidae, Simuliidae, Dixidae, Ephydridae, Tanypodinae and Muscidae, Blephariceridae, Chironomidae, Tabanidae, and Tipulidae (Archan et al. 2015; Subramanian and Sivaramakrishnan 2007).

The members of the order Diptera were found to be high in summer in comparison to winter and monsoon. The abundance of Diptera during summer was due to the presence of summer-related resources. During monsoon, the water volume increases, that diluted the nutrients of the water resulting in decrease in species

(Lonkar and Kedar 2013). Among Diptera, *Chironomous, Culex, Eristalis, Rhapidolabis, Tobanus, Musca autumnialis, Nepa, and Ranatra elongate* were used as a bioindicator in Ambazari lake, India (Lonkar et al. 2015).

4.2 Class: Crustacea

Crustaceans are intolerant of pollution. The taxa were represented by the family Palaemonidae. The abundance of the Crustaceans is known to present in high numbers at the time of monsoon and completely disappear in summer (Archan et al. 2015; Sharma et al. 2013). Various orders used as bioindicators are explained subsequently.

Cladocerans: Cladocerans are small freshwater animals (0.2–6 mm) also known as "water fleas." Cladocerans include four different orders that were recognised by Fryer (Fryer 1987). They include Anomopoda, Ctenopoda, Onychopoda, and monotypic Haplopoda. Most species of Cladocerans occur in continental fresh or saline waters. They have a significant role in the food chain of stagnant waters. Most species are filter feeders; Onychopods and Haplopods are predatory. The distribution of Cladoceran species does not depend on latitude rather on the temperature of water (Korovchinsky 2006). From northern India to East Asia, five genera and around 100 endemic species are found. Rivers those are often negatively influenced by human activities have decreased diversity, or extinction of some Cladoceran species are observed (Forró et al. 2007; Korovchinsky 2006). Thus, they are used as bioindicators. *Daphnia magna* is used as the bioindicator to monitor the pollution in Kathjudi River, Odisha, India (Rath 2015). In Hiranya Keshi, (Gadhinglaj, Maharashtra, India) four different Cladeocerans species like *Moina micrura, Ceriodaphni areticulate, Daphanosoma sarsi, Leptodora kindtii* were observed (Sakhare and Kamble 2014). In Maijan Bill (Assam, India), a wetland channelized with Brahmaputra River has 13.11% Cladocera found among all the planktons (Abujam et al. 2011). The Cladocera species includes in water bodies are *Bosmina longirostris, Ceriodaphnia* sp, *Daphnia sp.* (Abujam et al. 2011). *Daphnia magna* and *Monia macrocopa* are also used as bioindicators in the Han River, Korea (Kim et al. 2012). Cladocera forms a remarkable bioindicator for water pollution in Hiranya Keshi River, Gadhinglaj, Maharashtra (Sakhare and Kamble 2014).

Ostracods: Ostracods play an important role in determining the structure of water bodies (Lawrence et al. 2002). In these microenvironments, they are eaten by fish or gastropods. Ostracods compete with oligochaetes and amphipods for food sources (Modig et al. 2000). They are extremely sensitive to salinities, pH, water depth, and temperature (Modig et al. 2000). Thus, Ostracods are used as bioindicators to detect the quality of river water. In Hiranya Keshi River (India), the presence of a surplus amount of Ostracod species provides sufficient food for aquatic organisms. Two species, *Stenocypris* and *Cyprius subglobosa*, were found in the water bodies (Sakhare and Kamble 2014). *Cypris pubera along with Ilycypris inermis*, *Ilyocypris gibba*, *Tonnacypris lutaria, Eucypris virens* species were as a bioindicator to detect the water quality of Bolu River in Turkey (Külköylüoğlu 2004). Seed shrimps are used as biomonitors, and they are very sensitive to environmental changes such as industrial development, increasing population growth, and eutrophication (Ruiz et al. 2013). Ostracods also used a bioindicator for water pollution in Hiranya Keshi River, Gadhinglaj, Maharashtra (Sakhare and Kamble 2014).

Decapoda: In Ambazari Lake, Nagpur, India, Decapoda were used as bioindicators (Lonkar et al. 2015). In this study, *Paratelphusa aquemont* and *Gelasimus* sp. are used as bioindicators. Shrimps (*Caridina, Litopenaeus, Macrobrachium, and Neocaridina*) are analysed for Organochlorine pesticide (OCP) residues in six streams of Cauvery River, Karnataka, India (Apstein 1907). Mud crab *Scylla serrata* is an ecologically and economically important species and used as a bioindicator for Sundarbans Biosphere Reserve (Ali et al. 2004). Freshwater crab *Barytelphus aguerini* was used as a biomarker to Chlorpyrifos (Organophosphorus pesticide) exposure in the Gomti River (Tripathi et al. 2019). Two species of freshwater prawns, *Macrobrachium lamarrei* and *Macrobrachium dayanum*, are used as a bioindicator to detect the acute toxicity of nickel in Lucknow, India (Shuhaimi-Othman et al. 2011).

Amphipoda: Amphipods are already used as an excellent bioindicator of water (Contreras et al. 2010). Malacostraca fauna is carried out in the upper Crouch River Basin, Turkey. Concerning water quality, seven taxa have been identified. Five of them belong to G*ammarus sp*. They include *G. bristeini, G. balcanicus, G. fossarum, and G. kischineeffensis.* Besides that one, *Isopoda (*Asellus aquaticus*)* and *Decapoda (Potamonibericum)* were identified (Baytaşoğlu and Gözler 2018). Gammarus species are widely used as a risk assessment of water quality (Adam et al. 2010). The relation between acetylcholinesterase inhibition, feeding, and locomotor behaviours is investigated in adult male *G. fossarum* after exposure to the chlorpyrifos and methomyl (Xuereb et al. 2009) (Table 2.2).

5 Arthropods Used as Bioindicators in South Asian Rivers

South Asian Rivers are the last global frontiers, rich in biodiversity of threatened and endangered species. At the same time, these rivers nourish millions of indigenous people, since India is the largest country in South Asia. Besides this, the most threatened rivers are found in the Indian subcontinent (WCMC 2000) (Barman 2014; Karn and Harada 2001). Indian rivers are rich in aquatic insects, still, only a handful of studies have been carried out on the ecological role of aquatic insects. In India, the Western Ghat is the hotspot, and it is globally important for various freshwater species. In Western Ghat, 16% of (1146) freshwater taxa are at the risk of extinction. In Eastern Himalayan Hotspot, 1.9% of freshwater taxa were assessed as close to extinction (PV 2016). The extinction number is alarming and warrants the need for conservation. The use of the most diverse species is an easy way to monitor the health of the river ecosystems (PV 2016). Many studies have identified different taxa to monitor the health of the South Asian River. In India, 4000 species of

Table 2.2 List of crustaceans used as bioindicator

Order	Genus	Species	Used in which river	Level of pollution tolerance
Cladocera	*Daphnia*	*D. magna*	Kathjudi river, Odisha, India	Rath (2015)
	Moina	*M. macropa*	Han River, Korea	It is resistant to 108 and 31 ng/L of trimethoprim and chloramphenicol Choi et al. (2008)
Decapoda	*Scylla*	*S. serrata*	Estuary in Sundarbans, West Bengal, India	The presence of arsenic resulted in hypersecretion of the mucoid element Saha et al. (2010)
	Caridina	–	Cauvery River, Karnataka, India	It can resist beta-HCH pesticide residue (74.34 microgram/L) in shrimp *Caridina sp* Abida et al. (2009)
	Baryelphusa	*B. guerini*	Gomti	Crab shows restlessness; escape behaviour, increased oxygen consumption with an increase of pesticide concentration Srivastava et al. (2013)
	Potamon	*P. ibericum*	Upper Coruh River basin, Turkey	*P. ibericum* is rarely found in water having dissolved oxygen, temperature, PH, electrical conductivity, nitrite, ammonium, phosphate, and solid mixed water Baytasoglu & Gozler (2018)

caddisfly (Trichoptera), with a total of 50,000 species in the Oriental Region, and the genus Chimera (Philopotamidae) have 500 Southeast Asian species (De Moor and Ivanov 2007; Dudgeon 2002; Parey and Pandher 2016). This suggests insects can be used as a biological indicator to detect the water quality of the Rivers of India. Besides India, these species can be used to assess the pollution of other South Asian countries.

In South Asia, India is the largest country and rich in freshwater biota. In India, the most important river is the Ganga. Many studies represent the alteration of physicochemical parameters of the Ganga water and their changes according to loads of pollution. Besides Ganga River, studies are carried out on Cauvery River, Yamuna river, Tungabhadra River, Pravara River in Sangammer, Maharashtra, on the Cauvery River stretch in Karnataka, Kalpi, Gwalior, and River Gomati (Singh et al. 2010). All these studies reported the importance of aquatic arthropods on the nutritional chain of other organisms. Arthropods order like Trichoptera and Ostracoda are indicators of river water quality. The immature stages of these two orders have been used in toxicological studies. These diverse species are decreasing due to the overexploitation of river water and human activity (Singh et al. 2010). Inappropriate growth and loss of population of these groups are the signs of a disturbed ecosystem. Arthropods directly or indirectly influenced the health of the other organisms in an aquatic ecosystem (Akolkar et al. 1999). For example, Gammaurus of order Amphipod influences the colour and taste of the Salmon meat (Dehghani et al. 2014). This indicates to maintain the proper condition in the water ecosystem healthy population of these arthropods is needed. Less number of these arthropods in the water indicates a disturbed ecosystem (Archan et al. 2015). In River Basantar in the Jammu Kashmir, arthropods are the most dominant organisms. Order Diptera species were the most abundant in the polluted water. The insects from Odonata, Ephemeroptera, Coleoptera, and Hemiptera were observed in the lowest pollution site (Sharma and Saini 2016). The presence of these bioindicators detects human activities in Indian Rivers. Biological monitoring was also used to detect the water pollution in Kapila River (Sharma et al. 2013). River water pollution in Jharkhand was detected by various macrobenthos (Nesemann et al. 2017). Mukhopadhyay and Kumar used biological monitoring to detect the water quality of the river Yamuna using benthic macroinvertebrates (Kumar 2003; Panigrahi and Pattnaik 2019). The presence of various macroinvertebrates at the Hindon River in Uttar Pradesh detects the water health of the river (Kumar 2003). Various bioindicators are also used in four different regions of the Yamuna River to detect pollution (Amita 2011; Fouzia and Amir 2013). Plecoptera and Trichoptera are found in the Himalayan segment of the Yamuna River. Ephemeroptera is sensitive to pollution found in some regions of River Yamuna. In the Delhi region, Odonatan, Hemipteran, Coleoptera, and Diptera are observed in the Delhi segment of Yamuna River (Rani and Bhamrah 2014).

Some of the arthropods were less distributed with the pollution which includes Chironomidae, Hydrophilidae, and Noteridae; Lestidae Nepidae and Pleidae; and Hygrobidae and Hydrobiidae. Thus for these families, only their number gets affected (Kumar 2003). In Karamana River of Southern Western Ghat, the complete absence of some insect species occur in the most of the polluted region of the river (Bismi and Madhusoodanan Pillai 2013). A study on the river Beywa at Vidisha also reports arthropods as bioindicators in river water. The absence of some arthropods such as Cladoceran (a member of small crustaceans) revealed the worst state of the water (Patel and Datar 2014). The water quality of the river Kshipra in India was studied for water quality detection by using arthropods order such as Ephemeroptera, Trichoptera, Plecoptera, Coleoptera, Hemiptera, Odonata, Crustacea, and Diptera (Table 2.1). In different seasons different orders are used as biomarkers (Archan et al. 2015). Anthropogenic activity in the smart city Jabalpur was mentioned by Shukla et al. The insect order Odonata was used as a bioindicator to detect the level of water pollution in the Narmada River (Shukla et al. 2016).

Not only in India but in other countries of South Asia such as Sri Lanka various insect orders were used as biomarkers. The orders like Ephemeroptera, Plecoptera, Trichoptera, and Coleoptera are used to detect the impact of anthropogenic activity in rivers (Madhushankha et al. 2014). Godward River of Nepal was known to be dominated by four orders, Trichoptera, Diptera, Ephemeroptera, and Coleoptera, from Class Insecta (Vaidya 2019). Hydropsychidae is a family of Trichoptera and is used as bioindicators (Vaidya 2019). In Pakistan, few insects were used as biological indicators to detect the level of heavy metals in water. Insects used for this study

Table 2.3 Various characteristics that make arthropods especially insects a good bioindicator (Redrawn from Shrivastava et al. (2018)

Characteristics	Description
Diversity and species richness	Out of five species, four species of animals are known to be insects
Easy handling	Due to the small size of the insects, the capture and transport of the samples are easy
Ecological reliability	Many insects are sensitive to different abiotic factors, which decide the specific habitats for insects
Vulnerability to small changes	It allows the selection of the demographical and behavioural variables to present the habitat and which have a close correlation with the preselected area change
Organism's responses	Indicates the level of pollution in the water

Table 2.4 Various pollutants/factors affecting the survivability of arthropods

Chemical/pollutant	Invertebrate group	References
pH/acidification	Lentic chironomids	Mousavi (2002), Shrivastava et al. (2018)
Heavy metals	Caddisfly	Azam et al. (2015), Shrivastava et al. (2018)
Nitrogen and phosphorus	Lotic insects	Lemly and King (2000), Shrivastava et al. (2018)
	Lentic chironomids	Brodersen and Lindegaard (1997), Shrivastava et al. (2018)
Pesticides	Dragonflies	Shrivastava et al. (2018), Takamura et al. (1991)
Organic pollution	Cladocera	Shrivastava et al. (2018), Tatarazako and Oda (2007)
Coal mine runoff	Trichoptera	Fernández-Aláez et al. (2002), Shrivastava et al. (2018)

were a libellulid dragonfly (*Crocothemis servilia*), an acridid grasshopper (*Oxya hyla hyla*), and a nymphalid butterfly (*Danaus chrysippus*). Metal concentrations in insects were significantly higher near industries and nallahs in Gujarat, Pakistan. This indicates that these insect groups were potential indicators of metal contamination and can be used for biomonitoring (Azam et al. 2015).

The water quality of South Asian Rivers is monitored by using some groups of insects and crustaceans. The various characteristics that make the arthropods a suitable bioindicator are described in the table (Table 2.3). Among insects, the orders Hemiptera, Coleoptera, Trichoptera, Ephemeroptera, Odonata, Collembola, Diptera, Plecoptera, Orthoptera, Blattodea, and Megaloptera are used in several studies as potential bioindicators (Barman and Gupta 2015; Chandra 2011; Gogoi and Guha 2000; Sivaramakrishnan et al. 1996; Subramanian and Sivaramakrishnan 2007) (Table 2.1). Among them, Odonata is known to be the most sensitive to any kind of disturbances in their habitats (Barman and Gupta 2015; Shukla et al. 2016). It has been reported that butterflies and grasshoppers also have ecological importance since they are sensitive towards the stressors in their habitats (Shrivastava et al. 2018). Thus, these insects are successfully used in biomonitoring for environmental pollution such as heavy metal contamination in the river water of urban areas (Solomon et al. 2007) (Table 2.4). These insects also work as biogeochemical indicators and can be used as an important tool for the ecological evaluation of aquatic ecosystems. The concept of using organisms as bioindicators was introduced first by the European countries (Azam et al. 2015; Stankovic et al. 2014) and later accepted globally.

6 Advantages of Biomonitoring the River Ecosystem

Biomonitoring is a necessary way of detecting the state of pollution in ecosystems. However, the selection of appropriate techniques is essential for successful biomonitoring (Chutter 1972; Jones et al. 1981; Li et al. 2010; Persoone and De Pauw 1979). Altogether, 50 different methods have been developed for biomonitoring (Mandaville 2002). These methods are developed based on the characteristics of the bioindicators. Different methods include various parameters such as diversity indices, biotic indices, integrated biological indices, and multimeric, multivariate, and functional approaches. Recently, early warning biomarkers are gaining importance in biomonitoring. Bioassay protocols are widely used, which use macroinvertebrates to test the toxicity of contaminants in the river water. A perfect index is sensitive to all types of disturbances and can discriminate those disturbances appropriately (Karr 1991). Several indices can follow the rules. However, there are some indices with multimeric or biological integrity that focus on more variables. We are discussing some widely used approaches to monitor the ecosystem (López-López and Sedeño-Díaz 2015). They are as follows:

6.1 Diversity Indices

Diversity Indices is a traditional way of biomonitoring. It is used to detect the response of the community to environmental variation. This method uses three community structures, such as species richness, evenness, and abundance (Margalef 1951; Shannon 1948; Simpson and Norris 2000). The undisturbed and less polluted environment marks high diversity, even distribution, and a high number of individual species (Stevenson et al. 1996). It is the best method because changes in the species composition in the water indicate the pollution level. Now, these indices are preferred to be used with other metrics and called Multimeric Approaches (Li et al. 2010; Metcalfe 1989).

6.2 Multimeric Approaches

Multimeric approaches are the integration of a set of metrics, which indicate the functional and structural state of the ecosystem. This approach includes variables such as life cycle, richness, density, disease, dominance, relative abundance, life history, pollution tolerance, and functional feeding groups. All these variables will provide ideas regarding the anthropogenic disturbances in the river (Barbour et al. 1996; Barbour et al. 1999; Fausch et al. 1990; Karr 1981, 1986). This approach is widely accepted in the United States, and recently, it has been accepted by other countries (Barbour et al. 1999; Sivaramakrishnan et al. 1996; Thorne and Williams 1997; Vlek et al. 2004).

6.3 Indices of Biological Integrity

Indices of Biological Integrity or Index of Biotic Integrity (IBI) was first developed based on the assembly of fish (M. Barbour et al. 1996; Fore et al. 1996; Hughes and Oberdorff 1997; Karr 1986; Kerans and Karr 1994; Miller et al. 1988; Simon 1998). Later IBI was used for other organisms such as algae, macrophytes, macroinvertebrates, or other taxa. The value of each matrix depends on the comparison with a regional site that possesses no or little influence of disturbances (López-López and Sedeño-Díaz 2015; Weigel et al. 2002). Several factors like the number, richness, composition, and abundance of species are used for IBI. River Invertebrate Prediction and Classification System (RIVPACS) is a special case of IBI which uses multivariate approaches. It is a predictive approach to identify the alteration in macroinvertebrates with the response to human activities (Weigel et al. 2002).

6.4 Biotic Indices

The biotic index is a combination of relative abundance based on a certain group that is sensitive to a single index or score (Tolkamp 1985). The sensitivity or tolerance of an indicator species varies for several environmental parameters such as pH, organic pollutants, heavy metals, pesticides, and eutrophication. Some commonly used biotic indices are (1) Trent Biotic Indices, (2) Extended Biotic Indices, (3) Chandler's Score System,(4) Biological Monitoring Working Party Score (BMWP), (5) Average Score per Taxon, and (6) Hilsenhoff's Biotic Indices. Among all the indices, BMWP and its derivative, IBMWP are widely recommended by the Water Framework Directive and widely used in European Countries (Li et al. 2010; Metcalfe 1989).

6.5 Multivariate Approaches

These approaches help to detect the site-specific pattern of fauna (Wright 2000). Statistical analyses were first carried out in the absence of environmental stressors. Later, this evaluation was carried out in comparison with the expected fauna of that region. First, these approaches are carried out in Rivers of the United Kingdom. Some of the popular multivariate approaches are River Invertebrate Prediction and Classification System (RIVPACS), AusRivAS (Australian Rivers Assessment System), BEAST (Benthic Assessment Sediment), and ANNA (Assessment by Nearest Neighbour Analysis) (Linke et al. 2005; Niemi and McDonald 2004; Reynoldson et al. 1995; Rosenberg 1998; Simpson and Norris 2000; Wright 2000).

6.6 Functional Approaches

Necessary characterisation of the ecosystems is required for the structural as well as the functional approaches to know the ecological integrity. Some of the important functional approaches are (1) Functional Feeding Groups (FFGs) and (2) Multiple Biological Traits.

6.6.1 Functional Feeding Groups (FFGs)

FFGs are the main components of the River Continuum Concept (Gessner and Chauvet 2002; Vannote et al. 1980). It is used to detect the processes of ecosystem level in freshwater ecosystems. This approach is used in the form of a single feeding group such as absolute or relative abundance. The ratios between these two groups include the prospects of all the trophic levels. This approach is now combined with the other multimeric approaches (Barbour et al. 1999; Gayraud et al. 2003; Merritt et al. 1999; Pavluk et al. 2000; Rawer-Jost et al. 2000).

6.6.2 Multiple Biological Traits

Biological traits give information about the characteristics of habitat, biological, and ecological aspects of the species. The physical characteristics of the species such as size, life cycle, reproduction, and feeding habits are measured as environmental stressors (López-López and Sedeño-Díaz 2015). These approaches are used to monitor the level of pollution in river water. This approach is used in many European countries (Charvet et al. 1998; Gayraud et al. 2003; Pont et al. 2006).

6.7 Early Warning Bioindicators

The toxicity of the river water in an organism can be monitored at molecular, cellular, biochemical, physiological, morphological, and behavioural levels. Altogether, it affects the health of the individual organisms present in the river ecosystem.

Detection of toxicity can prevent the deleterious effect at population, community, and ecosystem levels (Hyne and Maher 2003; Sherry 2003). Biochemical approaches are depending on the presence or absence of biochemical parameters in bioindicators to detect the health of the ecosystem at an early stage (Bartell 2006). These parameters include oxidases, acetylcholinesterase, cellulose/carbohydrase, oxidative stress, genotoxicity, ion regulation, and stress proteins in various organisms (Barata et al. 2005; Bonada et al. 2006; Hyne and Maher 2003).

6.8 Bioassays Protocols

The bioassays are carried out to measure sentinel species ecological characteristics, feeding habits, respiratory modes, and it is placed in the food web. Macroinvertebrates have an important role in the food web of aquatic ecosystems. They can accumulate contaminants from the water or sediments. Thus, a contaminant can be detected as acute or chronic. These tests can be performed by using bioassays. The Environmental Protection Agency has developed several species to detect the toxicity of the contaminants in the freshwater ecosystems. Commonly used species are *Hyalella azteca* and *Chironomus tentans.* The endpoint for *H. azteca* is survival and for C. tentans is survival and growth (Borgmann and Munawar 1989; Gerhardt 1996).

6.9 Widely Used Methods of Biomonitoring in River

6.9.1 Measures of the BMWP Scores

BMWP score is calculated by adding all the individual scores of different families present in the river. Scores of different families reflect their ability to be tolerant of the ecosystem. Higher scores indicate the highly tolerant species in polluted water. Similarly, a low score indicates the less tolerant species present in the polluted water (Callanan et al. 2008; Ganguly et al. 2018; López-López and Sedeño-Díaz 2015; Subramanian and Sivaramakrishnan 2007).

6.9.2 BMWP-ASPT (Biological Monitoring Working Party Average Score Per Taxon)

BMWP-ASPT is the ratio of dividing the score with total scoring taxa numbers. High BMWP-ASPT indicates the cleanliness of the river water and high scoring taxa. Low BMWP-ASPT indicates the disturbed river ecosystem and less scoring taxa (Callanan et al. 2008; Ganguly et al. 2018; Subramanian and Sivaramakrishnan 2007).

6.9.3 Percentage of Ephemeroptera, Plecoptera, and Trichoptera (%EPT)

The combined percentage of three different orders Ephemeroptera, Plecoptera, and Trichoptera are also used as a measure of water pollution. This percentage is directly proportional to the clear water quality. Higher percentage indicates clear, less polluted, pollution sensitive, and fast-flowing rivers (Callanan et al. 2008; Ganguly et al. 2018; Subramanian and Sivaramakrishnan 2007).

7 Conclusion

The foremost objective of biomonitoring is to safeguard biodiversity. The protection of biodiversity is beneficial for local communities, exploiters, and conservationists. Biomonitoring indirectly helps to prevent water pollution that is harmful to humans (Jain et al. 2010). Biomonitoring indirectly helps to assess anthropogenic activities (Shrivastava et al. 2018). The economic progression and increased population in various South Asian countries including India cause a serious threat to biodiversity. This warrants various traditional approaches to protect biodiversity since it is expensive to measure the level of pollution in rural areas (Barman and Gupta 2015). Thus, biomonitoring is used by conservationists to detect the health of river water in rural areas. The identification of biomonitoring species in the river ecosystem is the easiest and less expensive technique. The detection of highly abundant biomonitoring species is the best way to detect the level of water pollution in the Rivers (Azam et al. 2015; Bohac 1999; Carroll and Pearson 1998; Jain et al. 2010; Li et al. 2010; Markert et al. 1999; Markert et al. 2003; Oertel and Salánki 2003a, b; Paoletti 1999; Shukla et al. 2016). In South Asian countries such as India, various insects have been reported in water bodies. Their number varies with the physicochemical parameters of water. Thus, the type of species used as bioindicators varies from state to state. For example, in Sikkim—5941, in West Bengal—5818 species, 5118 species in Meghalaya, and 4160 species in Uttarakhand are used as bioindicators of rivers (Chandra 2011). The alteration in an abundance of these species as well as groups is used as a bioindicator for the conservation of river health (Azam et al. 2015; Lemly and King 2000; Shrivastava et al. 2018). Thus biomonitoring is a healthy way to protect the water bodies present in South-Asian countries.

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Chapter 3
Anthropogenic Stress on River Health: With Special Reference to Kangsabati River, West Bengal, India

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Abstract Freshwater body like the river has immense importance in respect to human civilization. In the present scenario, human activities pose a serious threat on river health through different ways. We have studied to evaluate the consequence of human activities on the health of the Kangsabati River. Consequently, river flow diversion, pit formation, channel incision, soil erosion, abrupt sedimentation in few places of river bed increases the tendency of flood breaking the riverside dam during rainy season due to extreme water pressure at the middle and lower stream. Huge pressure is also evolved by the articulation of plastic goods and invasive *Eichhornia* sp. at the pillar of the bridge on the river. Water quality has been gradually deteriorated which in turn breaks down the chain of the river ecosystem particularly through habitat destruction, disturbance of planktonic community, diminishing of fish diversity, eutrophication, the origin of Coliform bacteria, sporadic development. Indirectly, an unhygienic environment hampers the human health, public life system, and stress on economy. In a nutshell, a horrible situation emerges. This study enlightens the present situation of the concerned river in view of anthropogenic activities and it will play a directive role to understand the stress on the watery

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environment of Kangsabati River and to make a scientific plan for fruitful supervision to facilitate continuous progress.

Keywords Kangsabati river · River health · Anthropogenic stress · Water quality · River bed

1 Introduction

Rivers play an important role in the distribution of water resources from nature to human needs like irrigation, hydropower generation, fisheries activities, sand mining, dam construction, bridge construction, waste discharging, goods transportation, aesthetic pleasure, water supply, etc., but in the twentieth century, its excessive use reached to a level of stress which deteriorates the health of the river. River's physical condition determining factors like quality health, biota strength, health of land submerged by floodwaters, health of waterway, catchment health, health of the water current are used worldwide to assess the ecological fitness (Meyer 1997).

River's physical condition determining technique derived from the environmental course is widely accepted, much better, and dependable than the conventional technique (Ma et al. 2019). Presently, bioindication and biomonitoring are considered as promising ways to study the health status of water bodies (Parmar et al. 2016). In this context, appraisal of health of large river was also done by fish population as bioindicator particularly *Xenocypris davidi and Hemibarbus labeo* analyzing characteristics of metrics of standard length, average age, fecundity, weight and gonado-somatic index (Chen and Jia 2013).

We know that river health is strongly correlated with the river ecosystem. Currently, biomonitoring approaches and bioindicators are used to evaluate the environmental health of the stream and rivers. Periphyton, benthic macroinvertebrates, and fish are the most common indicators in river biomonitoring (Li et al. 2010). In addition, Diatom algae can reflect the average status of the water environment as well as contamination. *Nitzschia communis*, *Navicula minuscula* can indicate eutrophication of water whereas *Synedra ulna* can indicate the zinc concentration in water (Li et al. 2012). Not only that but also the growth tendency of Diatom and the morphological changes by the mechanism of biosorption and bioaccumulation indicate heavy metal pollution (Ding et al. 2012).

River ecosystem provides notable aesthetic facilities for the purpose of pleasure such as fish catching, swimming, boating, sports, tenting, stepping out, etc. Numerous studies suggested that fish species richness is directly correlated with fishing activity and is considered as a powerful marker of biodiversity in the ecosystem of a watery environment (Heino et al. 2005) and ecosystem health also (Rapport et al. 1998).

Human interventions cause the diminution of various plants and animals of a river (Manny and Kenaga 1991). So many researchers suggested that loss of biodiversity is reflected in ecosystem function namely rate of production, biogeochemical cycling, and ecosystem characteristics like stability being declined (Tilman et al.

2006; Balvanera et al. 2006). The functions of ecosystem are more significant to human civilization, and ecosystem services (Millennium the Ecosystem Assessment 2005; Kareiva et al. 2011). In respect of human intervention, Mittal et al. (2015) documented changes in river flow.

Flood plain health includes the impact of vegetation to soil erosion shape or soil of the bank, its height and width, changes due to chemical, pesticides; changes due to flood dynamics. River channel health comprises the length and width of the river water flow path, lengthwise connection, impact of dams and weirs, conditions of plants and river. Furthermore, flow health increasing ecological motion does not essentially signify upliftment in fish dwelling state (Ceola et al. 2018).

In India, based on geographical location and origin rivers are the two types—(1) Himalayan river and (2) Peninsular river. Himalayan rivers are remarkable for habitat heterogeneity regarding the colonization of aquatic fauna. Furthermore, the delicate state of the soil, removal of forests of riparian habitat, soil erosion in the upper segment is widespread causing the critical situations to downstream sedimentation following habitat demolition (Srivastava 2007). Ganga, Brahmaputra, and Indus river systems are the principal Himalayan Rivers. On the contrary, the entire peninsular river is absolutely dependent on rainfall particularly the monsoon period which steers ecological and biological condition. Peninsular Kangsabati river is presently now under severe stress on account of human activities. In addition, many workers previously reported that most of the Indian rivers are undergoing threatening caused by anthropogenic sources (Trivedi 1988; Gopal and Sha 1993, Patra et al. 2017).

In connection with this investigation, we analyze the prolonged observation of human activities which stress on river health, flood plain health, river channel health, quality health, and catchment health. Specially we evaluate the impact of stressors comprising sand mining in different areas, construction of the dam, unscientific construction of riverside and bridge, set up of brick factory adjacent to the river, expansion of agricultural field and its surface runoff, overfishing, destructive fishing, irrigation, recreational use, mass bathing, and religious activities, domestic sewage discharge, urbanization, etc.

For an effective management of anthropogenic stress on the Kangsabati river, a number of techniques like modeling approaches and monitoring programs, field surveys have been introduced. To mitigate the stress on river health and better checkup, the concerned authorities are to be empowered for formulating and managing a number of effective scientific inferences in the form of strategy and policy. Special consideration should be administered to the river ecosystem when at risk to several troubles like contamination, eutrophication by chemical fertilizer and pesticides being surface runoff from agricultural fields, etc. Water resources monitoring after the regular intervals for the qualitative assessment is urgent where it is supplied for domestic use, healthy ecosystem, and hygienic environment. Clarifying the emerging situation of the river water body express the disturbing condition to the administration, decision-makers and the common people in a serious and logical approach for proper management in future prospect.

2 Materials and Methods

Kangsabati river is a freshwater body having a length of about 465.23 km (Fig. 3.1). Originating from the Chota Nagpur plateau of the Jharkhand state of India it flows along the Purulia, Bankura, and Paschim Medinipur districts in West Bengal and finally, it empties into the Bay of Bengal. Particularly, it runs along Jhalda, Purulia (Purulia district); Khatra, Ranibandh (Bankura district), and Binpur, Kharagpur, Midnapur, Keshpur, Daspur region (Paschim Medinipur district). Then the river divides into two branches at Bhairabbanki (Keshpur region). The northern branch flows through the Daspur area touching Palashpai and finally meets in the Rupnarayan River. The other branch in the south-east direction combining with the Keleghai river opens into the Haldi River which proceeds to the Hooghly River at Haldia.

The key points of the healthy river comprise an enriched diverse community of organisms and a supportive ecosystem. In addition, the health of a river is controlled by various factors viz. maintenance of its composition and function; recovery after interruption; support to surrounding biota including human population; regulation of key processes like sediment transportation, nutrient cycling, and energy interchange. However, in view of ecological and human value river health is environmentally significant.

Environmental Impact Assessment (EIA) has been carried out to investigate the activities associated with mining and processing of sounds i.e. effect on Kangsabati

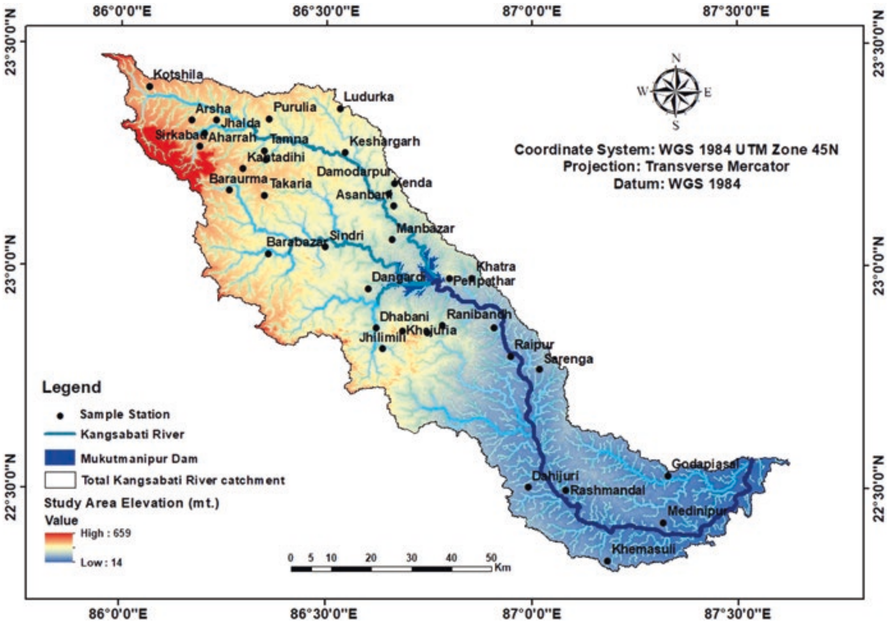

Fig. 3.1 Location of the study river Kangsabati, West Bengal (India)

river health ecosystem and its bank erosion. River Health Assessment (RHA) tools are used for the assessment of the ecological fitness of the river like catchment health, flow health, flood plain health, channel health, quality health, and biotic health indicators. In order to assess the river health, we emphasize on following two key components, i.e., quality health and biota health (Table 3.1).

1. Sand mining—Field survey cum investigation data have been collected at regular intervals along the river and direct interviews with the local people as well as workers have been done.
2. To better understand the anthropogenic stress on river health with special reference to the Kangsabati river, we explore the importance of bioindicator species. Indicators—phytoplankton and Zooplankton qualitative analysis. The planktons were collected by plankton net from three points of Kangsabati river viz., tributaries of Kangsabati reservoir, Kangsabati river, and Kumari river at Mukutmonipur; Mohanpur bridge at Midnapore and Kolmijore at Daspur. Then

Table 3.1 Indicators of river health quality

Quality health	---- > Water quality ---- > Discharge of pollutants ---- > Bioindicators ---- > Eutrophication
Biota health	---- > Flora, fauna, and their habitats ---- > diatoms ---- > macro/micro invertebrates ---- > fish ---- > transport and transformation of nutrients
Catchment health	---- > land use pattern ---- > catchment disturbances ---- > tributaries ---- > Forest cover
Flow health	---- > impacts of dams and pillars of bridge on river bed ---- > discharge/velocity/regime ---- > water extraction structures (e.g., pumps) ---- > ground water level ---- > longitudinal connectivity
River channel health	---- > river bed forming ---- > sand mining ---- > fluvial geomorphology (sediment transport/erosion) ---- > physical form—Channel bed form/material ---- > length and width of the river
Flood plain health	---- > lateral connectivity ---- > use of chemicals of floodplain farming ---- > riparian vegetation ---- > geomorphologic complexities of the river like ripple, pools, sandbars ---- > stream variability in terms of course/channel shifting ---- > Bank erosion and slope angle ---- > floodplain soil organic/inorganic matter ---- > ecological status of floodplain wetlands

100 mL of water sample was collected from each sampling point at 3 months regular interval through the filtration of 100 L of water. Then it was preserved using 5% formaldehyde solution in a small plastic container. Subsequently, supernatant fluid was removed and bottom sedimentary fluid was analyzed by high-resolution microscope. Zooplankton species identification was done following the key, standard literature, and authenticated monographs of Needham and Needham (1962); Sharma (1998) whereas identification of phytoplankton was done under the microscope following standard books, keys, and literature of Prescott (1978), Philipose (1967), and data available from www.algaebase.org

3. Fish catch data with the help of fishermen have been observed and identified following taxonomic keys, monographs, and standard literature of Talwar and Jhingran (1991); occasional paper no 220, Zoological Survey of India, Kolkata (Mishra et al. 2003) and also with the assistance of fisheries expert, Zoology Department, Vidyasagar University, India.
4. Comparative data of water quality of Kangsabati river have been incorporated from the Source—West Bengal Pollution Control Board, Water Quality Information System, Medinipore (W): www.wbpcb.gov.in).
5. Questionnaire: Twenty questions had been framed for the households residing close to the bank of the Kangsabati river and the workers in Brick Kilns. The questions include the condition of the environment and the nature of the area. The people who are specialists in the brick making were also interviewed. Observation of the nature of the site was also done.

Agricultural practice, construction of a dam, mass bathing, and other related information were evaluated by direct interviews to the local people.

3 Results and Discussion

3.1 Impacts on Riparian Areas

There are so many human activities that cause major alteration to riparian areas. It involves the changes in hydrology, geomorphology, and vegetation (Table 3.2).

3.2 Sand Mining

To categorize the type of sand mining and assess of geomorphic influence by reason of random and unempirical sand mining, we select river in between Gandhi Ghat to Temuhani Ghat, West Bengal. By means of geographic information system (GIS) and field survey, we made an effort to recognize the effect of this massive pulling

Table 3.2 Anthropogenic impact on riparian ecology

	Human activities	Direct impact	Indirect impact
Geomorphic	---- > sand mining ----- > construction of dam ----- > construction of pillar based bridge ----- > unscientific riverside construction ---- > construction of brick factory Adjacent to river ---- > formation of distributaries (canal) to divert the flow of flood water	1. During the rainy season, high level of water creates tremendous pressure on river bundh. 2. Flood. 3. Soil erosion. 4. Dumping of bricks within the river beds making bed hard. 5. Sedimentation. 6. Increase streamflow along a particular face	1. Alter vegetation species. 2. Alter wildlife habitat. 3. Alter aquatic habitat. 4. Water level fluctuation. 5. Natural behavior of rivers changed. 6. Harvested croplands being submerged underwater-rotten/ destroyed in a particular flood area which impacts on economy indirectly. 7. Affect the rugosity of Kangsabati river
Hydrologic	---- > mass bathing and religious activities ----- > over fishing ----- > destructive fishing applying chemicals ----- > washing using detergent and soaps ----- > dead body of animal throwing Into the river ----- > combustion of the dead body of human at the embankment of River	1. Coliform bacteria increases. 2. Fish assemblages reduce in size. 3. Fish community decline. 4. Eutrophication. 5. Water toxicity and destruction of other aquatic organisms	1. Water quality deteriorates. 2. Fishermen community and fishing activity. 3. River ecosystem tends to break down. 4. Rotting of submerged macrophytes particularly in the littoral zone
Agriculture	---- > use of excessive chemical Fertilizer ----- > indiscriminate application of pesticides ----- > expansion of the agricultural field ----- > excavation of agricultural land for Brick making ---- > river water supply for irrigation Purpose through pump machine	1. Being surface runoff mix to water body make toxicity. 2. Critical situation for aquatic organism. 3. Morphological alteration of riparian areas. 4. Agricultural land degradation for the brick factory in Kangsabati river basin. 5. Water level reaches to the lower level	1. Soil quality degrade. 2. Soil erosion. 3. Hampers river ecosystem

(continued)

Table 3.2 (continued)

	Human activities	Direct impact	Indirect impact
Recreation	---- > picnic at the close proximity of River ----- > oil engine machine boating for Public transport from one side to another of river ----- > motorized boating for tourists at Mukutmonipur dam ----- > bomb bursting at watery environment during God Sitala Utsava	1. Plastic and thermocol pollution. 2. Water and noise pollution. 3. Threatening of aquatic organisms. 4. Reduced ground cover and physical damage to roots	1. Fish death, bioaccumulation of plastic components. 2. Water clogging. 3. Degradation of streambank. 4. Sedimentary particles negatively impact on water-living organisms. 5. Surface water unsuitable for plankton
Rapid urbanization and industrialization	---- > road construction and extension ---- > water despoliation/ abstraction from river setting pump machine on river embankment for supplying urban or municipal area. ---- > domestic and municipal waste dumping adjacent to riverside near Midnapore and other towns	1. Groundwater levels gradually become low during summer. 2. Removal of vegetation. 3. Sewage pollution. Increase water pollutants	1. Decrease groundwater recharge. 2. During summer agriculture and normal public life disturbs. 3. Alter water pathways to stream
Others	---- > sand loading, transport and unloading by oil engine machine boat ----- > sand loading also directly by Lorry in the river bed in few places of the river along upstream ---- > introduction of exotic fish species (at the tributaries of Kangsabati river, Kangsabati reservoir and Kumari river) ----- > introduction of invasive plant species like *Eichhornia crassipes* ----- > in dry season fecal matter discharge by rural people on river bed.	1. Water pollution by leaching oil. 2. Disruption of the river bed. 3. The population of small fishes to be reduced being eaten by exotic fishes. 4. During flood increases water pressure at the bridge pillar and lock gate point	1. Stress in native fish species and vegetation

out of the sand on this river. We have pointed out decaying of riverbank, unsteadiness of river bars, channel splitting up, and pool-riffle sequence alteration which are the main geomorphic impacts of sand mining activities (Fig. 3.2c).

The hydrology and morphology are now partially controlled by human activities, wide-ranging removal of sand from the Kangsabati river bed was started since 1999 due to increasing urbanization, industrialization, factory and construction, and earning extra money.

Mainly lower part of the river is alluvial rich but it is affected by uncontrolled sand mining. Bhattacharya et al. (2019) found in sand mining places where physicochemical parameters change the diversity index, species richness. Other than that Bhattacharya et al. (2015) reported that quarrying a pit of Kangsabati river creates nick points on edge of upstream and makes a trap on the river bed. Consequently, interruption in the flowing water along the downstream and sediment transportation has been focused. Insufficient sediment load in the downstream particularly in between Mohanpur and Singaghai during the rainy season due to a high volume of water increases the viability of flood. They revealed that mining activities on the Kangsabati river have a severe impact on river dynamicity as well as hydrological responses. In a word, flow separation (Fig. 3.2a), channel incision, and sandpit are the major impacts of sand mining. Finally, it has been found that river bed level is lower in the downstream than that of up and midstream whereas channel incision increases towards up and middle stream. Overall, shifting of nick point in the downstream increases flood tendency altering pool and ripple. Sometimes they use pump machines to collect and store huge sand. Uncontrolled sand extraction along rivers evolves ecological disaster. It is evidenced strongly in the Kangsabati river. This river bed is rich in molluscs—*Lamellidens* sp. and *Lymnaea* sp. On account of too much sand pulling out the clam bed became invisible, i.e., habitat destructed. In a straight line manner sand mining influence the physical processes as well as biological communities (Saviour 2012). Ghosh et al. (2016) also gave a brief account on geomorphic modification in the Damodar river for the sake of sand mining.

3.3 Agricultural Practices

Agricultural land at the riparian zone of the river causes the physicochemical properties to change of water. Large lands along the river are used for agricultural practices where chemical fertilizer and pesticides are applied. Subsequently, it being surface runoff by rainwater mixes to the river water body and cause water pollution which in turn it effects on aquatic life. Pertaining to these Gowda et al. (2016) revealed that the water of the Cauvery River was more or less contaminated by human activities, for instance, agricultural practices, and urbanization. Improper land or waste disposal particularly municipality areas like Midnapore and few adjacent semi-urban places Khukurdaha, Singaghai, Temuhani, Kolmijore, and others lead to the destruction of the Kangsabati river ecosystem. A similar type of observation was also reported by Oladeji and Adelowo (2016). According to Hajihashemi

Fig. 3.2 (**a**) Kangsabati river flow changes due to sedimentation, (**b**) Construction of dam on river for the purpose of irrigation (**c**) Sand mining by local people, (**d**) construction of bridge on river bed, (**e**) oil engine machine boat used for excavation of sand, (**f**) construction of anicut on river for the purpose of irrigation

et al. (2020), water pollution may interrupt photosynthesis and so affects the ecosystem. Ekka et al. (2019) also researched on the effect of human activities on river ecosystem services in India and showed the result as a cause of pollution.

3.4 Construction of Bridge on River

Construction of bridges on river water has some structural influence (Lane 1955) on it. During monsoon, water flow increases suddenly. This time moderate amount of soil material adjacent to the pillar of the bridge wash away or remove by turbidity currents (Pasiok and Stilger-Szydio 2010). In the case of the Kangsabati river, four bridges have been found where such type of situation arises (Fig. 3.2d). The river changes its natural behavior.

3.5 Brick Factory Adjacent to River

There are so many Brick factories near the riverside as well as riparian zone particularly at Hatihalka adjoining area, Midnapore town. The smoke of Chimney not only pollute the air but also the dust of factory being washed off mix to the river water. As a consequence, turbidity increases, dissolve oxygen level decreases, and habitat of aquatic organism some time reach to the critical condition for their survivability. Abdalla (2015) investigated and found out the strong environmental effect of red brick production on the bank of Blue Nile at Soba West, Khartoum, Sudan. The author highlighted that the adjacent river has been attacked by the introduction of brick dust, flaming of different gases, and plants by heat. Smoke particles generated through factory Chimney deposited on leave particularly plant respiration and photosynthesis may be hampered. Furthermore, red brick materials threw into the river water. The contaminated water affects different aquatic organisms especially fish and also the whole ecosystem. Chakraborti et al. (2014) analytically proved that brick manufacturing industries were responsible for the pollution of different aquatic bodies. Molla (2011) revealed that brick factories create environmental problems in the embankment of river Ajay, India and focused on choking of watery environment of the concerned river and pressure on the riparian habitat. Dey and Dey (2015) observed that brick industries in different parts of Cachar district, Assam were responsible for environmental troubles like soil degradation, atmosphere pollution, water quality deterioration, and loss of biodiversity. Additionally, it causes ecological inequality and disorder in the hydro-geo-chemical and hydro-biological cycles that unfavorably change the food chain and food web of brick kiln-encompassed aquatic bodies. Das and Bandopadhyay (2012) showed that one of the causes of the flood is the illegal construction of riverbank near brickfields in Bangaon block in Ichhamati river. This literature strengthens our survey and analytical report.

3.6 Recreational Use

We have enlisted the recreational uses viz. at the tributaries of Kangsabati river, Kangsabati reservoir, and Kumari river, motorized and nonmotorized boating for the purpose of tourism; fish catching in groups throughout the year except fish breeding period; organizing a picnic in the picnic spot; water bomb bursting for entertainment during the communal ceremony like Goddess Shitala Puja; boat racing competition during on account of the local festival at the bank of the river in winter season; from upstream to downstream fish catching by rural people especially by Jele community; passengers shifting from one side to another by machine boat, etc. Such type of anthropogenic pressure damages the river health. Doi et al. (2013) estimated the effects of biodiversity, habitat structure, and standard of water on amusement indicating fishing, playing, walking, engaging in sports near the river, boating. Finally, they investigated a relationship between a decline in biodiversity and ecosystem health decreasing in recreational use. In a case study, Ingole (2002) reported the adverse effect of tourism on Miramar beach fauna in Goa. In connection with this, mass bathing near sea beach, construction, excessive vehicle, pedestrian traffic are the stressors in the coastal belt. As per the technical report of Andereck (1995), water bodies used for recreational purposes have augmented heaps of sediments, nutrients like nitrogen and phosphorus, microorganisms, pesticides, and petrochemicals either directly or indirectly. On the other hand, motor-driven boats and watercrafts create stream hazards which act as an inducer of water and sound pollution, riverbank erosion, and sediment suspension. It is not fruitful for aquatic life (Garrison and Wakeman 2000). Pollution, changing and demolition of natural homes, trapping, and fishing, spreading of diseases or animals for entertaining purposes can all cause declining of native wildlife populations in riparian areas and the adjacent water bodies (Knight and Gutzwiller 1995; Cunningham 1996). In complying the various data, we highlight similar opinions.

3.7 Harmful Fishing

Application of herbicides, chemicals in water is used to capture huge fishes at a time. It may cause disruption of photosynthesis in aquatic plants and coming out a dangerous situation for other aquatic organisms. Thus it affects the ecosystem damaging productivity and breaking the food chain.

3.8 Construction of Dam

Numerous temporary dams are constructed along the Kangsabati river for the purpose of irrigation which alters the riverine ecosystem to a lentic ecosystem (Fig. 3.2b). The creation of dam and water diversion systems block migration routes for fish and disrupts habitat excessively. Shah et al. (2016) mentioned that part of human intervention like the construction of dam also impacts on diverse flora and fauna of a river. Moreover, Kumar and Jayakumar (2020) focused on the impact of man-made actions like the building of a dam on the Krishna river applying flow health method founded on monthly average flow, timing, period and occurrence, etc. They indicated that low-current seasons exhibit remarkable positive alterations and high-current seasons exhibit negative alteration from the point of natural flow view. They clearly stated that hydrological alteration impacts aquatic and riparian habitat as well as key species of the river and adjoining atmosphere. Jeppesen and Sondergaard (2007) attributed the impacts on fish distribution and connectivity of fish population in view of the construction of a dam. Mittal et al. (2014) revealed that flow pattern attractions occurred owing to human activities and climatic changes in the Kangsabati River, India. They highlighted that river flow variability has been notably diminished as a result of the construction of a dam whereas low flows during non-monsoon months enhanced. On the other hand, they also mentioned that streamflow becomes reduced in monsoon by way of minor changes in non-monsoon streamflow. They analyze that the combined effect will lessen flow inconsistency, potentially influencing the ecosystem.

3.9 Introduction of Exotic Fish and Plant Species

At the tributaries of Kangsabati river, Kangsabati reservoir, and Kumari river, five exotic fish species viz. *Cyprinus carpio, Oreochromis niloticus*, *Oreochromis mossambicus, Ctenopharyngodon idella,* and *Hypophthalmichthys molitrix* are observed. Kar et al. (2016) tabulated 46 fish species in the Kangsabati river, while 39 fish species were recorded by Bera et al. (2014) in Kangsabati reservoir. Dwivedi et al. (2016) reported that *C. carpio and O. niloticus* are both exotic fish species that are harmful to fish biodiversity and a large number of indigenous fish species in the Yamuna river. They are also opined that the high percentage of abundance, distribution, and landing of such exotic species due to poor water quality. Bhakta and Bandyopadhyay (2007) also reported eight exotic species in the Churni river. In fact, benthic siluroid fishes are very good markers of environmental dilapidation (Wootton et al. 1996). Extreme low flow and high flood pulses affect the life cycle of benthic siluroid fishes by interrupting different stages like spawning, egg, hatching, rearing, and migration to the flood plains, upstream or downstream for feeding and reproduction (Poff et al. 1997). Besides, Mishra et al. (2009) documented that degradation of fish habitat caused by channel bed sedimentation may be associated with hydrological changes. Apart from this, *Eichhornia* sp. and *Hydrilla* sp. are recognized as exotic plant species in this river at the time of flood obstructing river flow. Overall, exotic species can decrease fish yielding and harm native organisms.

3.10 Fisheries Sector/Overfishing

The fisheries have altered hurriedly during the most recent decades because of modern technological advancement and extension. The overgrowing population and requirement for food generate the additional force on the fishery sector. Overfishing changes the configuration of fish assemblages (Ansari et al. 2006). In the tributaries at Mukutmonipur, it has been observed that *Labeo calbasu* and *Oreochromis niloticus* are gradually reduced in number. It may be due to overfishing and low reproductive rate.

3.11 Discharge of Plastic and Domestic Sewage

Plastic packets and other products containing in the municipality wastes being surface runoff come into the river water because garbage is unloaded at the riverside. Tourists during boating throwing the food packets made up of plastic into the river. Domestic sewage containing plastic packets through rainwater passes small drain to large drain which opens into the river. It clogs not only the water body but also adversely affects aquatic life and their habitats as well as higher organism through the food chain by means of bioaccumulation and biomagnification. We know the chemicals like polyvinylchloride (PVC), bisphenol A(BPA), diethyl hexylphthalate (DEHP) are the constituent of various types of plastic goods. Ram Proshad et al. (2018) assessed the poisonous effect of plastic goods on the physical well-being of human and environment. In addition, Ansari and Matondkar (2014) pointed out the anthropogenic activities causes pollution like plastic, oil, domestic sewage, pesticide which triggers on marine environment affecting fish habitats, growth, breeding, and overall threats to marine biodiversity. In this context, Ratemo (2018) studied on Athi River in Machakos country, Kenya, and found out that river flow had been changed due to industrial effluents. It was confirmed by water quality analysis, correlation, regression-based analytical results. In case of the Kangsabati river, it is also noticed that river flow is changed due to small industrial effluents discharge in nearby Midnapore and Kharagpur. Menon and Pillai (1996) also found out that large inputs of domestic waste and industrial effluents have burdened the river environment. Human activities like stressors reduce fish catching and biodiversity at numerous hotspots. In case of the coastal environment, anthropogenic activities lead to habitat loss in various ways. In connection with the mangrove ecosystem of Akkulam Veli lake, Thiruvananthapuram, South India, Jayadev et al. (2013) worked out that activities like domestic sewage discharge and industrial effluents deplete the diversity of fishes.

3.12 Cattle Washing

The bathing of cattle was performed by rural people adjacent to the Kangsabati river several times in a year. Water quality can be deteriorated due to microbial development in the water body. It is evidenced by the total count of coliform bacteria

(Table 3.3) Venkateswarlu (2014) also mentioned cattle washing was a cause of water quality degradation because of sporadic development in and around Musi River, India.

3.13 Soaps and Detergents

During mass bathing and washing of cloths, most of the people use soaps and detergents for their everyday life. Continuously, excessive use of it impacts on water quality meanwhile eutrophication. Excessive growth of harmful algae consumed by other aquatic animals may direct to breathing trouble and outbreak of fish diseases which can cause decline of fish as well as plankton diversity. In relation to this development of aquatic organisms may be obstructed tremendously. Apart from this, a high concentration of NaCl in water may kill animals and plants. It is also revealed by Kushwaha and Agrahari (2016). They observed the human activities which lead to the addition of a ton of soaps and detergents as anthropogenic pressure on the Tapti River at Gorakhpur city, India. The result revealed that P^H, free CO_2, chloride, hardness, carbonate, bicarbonate, COD, BOD, sulfate, nitrate, TDS, phosphate value increased significantly whereas DO decrease significantly. Similarly, we have also seen similar types of results (Table 3.3).

3.14 Mass Bathing and Religious Activities

Mass bathing of rural people and pilgrims during religious festivals as well as activities like Makar Sankranti, Chhot Puja, Tarpan directs to raise nutrient concentration, organic and inorganic substances in the watery environment together with diverse infective microbes which remains stable for a long time. So many reasons are also recognized but particularly mass bathing and religious activities are pointed out for the sudden increasing level of Coli form bacteria (Table 3.3). Experimental data indicates Kangsabati river water is under mild pollution. Similarly, bacteria like *Pseudomonas aeruginosa* which acts as biological indicator species is studied by Bhasin et al.(2015) in the Kshipra river, India. In addition, Bhatnagar et al. (2016) clearly depicted the result following the study of eight water bodies in Hariyana, India that water parameters like nitrate, alkalinity, biological oxygen demand, conductivity, total hardness, turbidity, total dissolved solids, coliform bacterial count were significantly amplified subsequently to group bathing and pious behaviors. The index of water quality confirmed the fact of deterioration.

Table 3.3 Comparative data of water quality of Kangsabati river (Source—West Bengal Pollution Control Board, Water Quality Information System, Medinipore (W): www.wbpcb.gov.in)

Parameters	Unit	2011	2013	2015	2017	2019
P^H		Min.—6.48 Max.—8.18 Avg.—7.63	Min.—6.97 Max.—8.34 Avg.—7.62	Min.—7.51 Max.—8.41 Avg.—8.01	Min.—7.10 Max.—9.29 Avg.—7.62	Min.—6.99 Max.—8.63 Avg.—7.43
DO	Mg/L	Min.—5.8 Max.—11.0 Avg.—8.2	Min.—7.2 Max.—11.1 Avg.—8.8	Min.—6.80 Max.—10.6 Avg.—8.71	Min.—4.20 Max.—12.40 Avg.—8.65	Min.—2.05 Max.—6.30 Avg.—3.91
BOD	Mg/L	Min.—1.10 Max.—6.10 Avg.—2.55	Min.—1.40 Max.—2.80 Avg.—2.15	Min.—1.0 Max.—3.20 Avg.—2.22	Min.—1.20 Max.—9.90 Avg.—4.98	Min.—2.05 Max.—6.30 Avg.—3.91
COD	Mg/L	Min.—1.90 Max.—17.3 Avg.—6.34	Min.—2.7 Max.—12.6 Avg.—7.13	Min.—5.33 Max.—13.52 Avg.—10.27	Min.—8.40 Max.—8.01 Avg.—15.70	Min.—7.06 Max.—20.99 Avg.—12.14
Total alkalinity	Mg/L	Min.—50 Max.—90 Avg.—71.27	Min.—56 Max.—122 Avg.—77.83	Min.—42.06 Max.—102 Avg.—72.28	Min.—62.19 Max.—108.0 Avg.—82.44	Min.—64 Max.—100 Avg.—80.07
Chloride	Mg/L	Min.—9.42 Max.—18.35 Avg.—11.93	Min.—5.82 Max.—17.3 Avg.—12.57	Min.—10.29 Max.—33.05 Avg.—18.14	Min.—6.65 Max.—161.51 Avg.—26.55	Min.—17.61 Max.—27.99 Avg.—23.44
Phosphate–P	Mg/L	Min.—0.01 Max.—0.19 Avg.—0.085	Min.—0.01 Max.—0.13 Avg.—0.058	Min.—0.02 Max.—0.09 Avg.—0.037	Min.—0.04 Max.—0.11 Avg.—0.07	Min.—0.03 Max.—0.54 Avg.—0.139

(continued)

Table 3.3 (continued)

Parameters	Unit	2011	2013	2015	2017	2019
Nitrate–N	Mg/L	Min.—0.04 Max.—0.58 Avg.—0.16	Min.—0.01 Max.—0.38 Avg.—0.155	Min.—0.01 Max.—0.16 Avg.—0.05	Min.—0.32 Max.—0.66 Avg.—0.52	Min.—0.02 Max.—1.49 Avg.—0.46
Conductivity	μs/cm	Min.—114.8 Max.—232.0 Avg.—198.25	Min.—84.4 Max.—242.0 Avg.—184.16	Min.—104.0 Max.—283.0 Avg.—189.43	Min.—143.0 Max.—799.0 Avg.—268.05	Min.—180.0 Max.—322.20 Avg.—240.6
Total hardness	Mg/L	Min.—54 Max.—128 Avg.—85.53	Min.—46 Max.—88 Avg.—72.5	Min.—41.58 Max.—100 Avg.—74.08	Min.—56.56 Max.—121 Avg.—77.72	Min.—66.00 Max.—90.90 Avg.—76.49
Total coliform bacteria	MPN/100 mL	Min.—9000 Max.—1,600,000 Avg. 286,916	Min.—4000 Max. 300,000 Avg.—76,916	Min.—5000 Max.—90,000 Avg.—22,166	Min.—11,000 Max.—90,000 Avg.—35,584	Min.—4900 Max.—35,000 Avg.—16,958
TDS	Mg/L	Min.—52 Max.—226 Avg.—139.8	Min.—50 Max.—158 Avg.—116.83	Min.—6 Max.—204 Avg.—132.66	Min.—54 Max.—469 Avg.—155.5	Min.—78 Max.—368 Avg.—173.83
Turbidity	NTU	Min.—3.47 Max.—427 Avg.—95.60	Min.—5.23 Max.—372 Avg.—88.79	Min.—7.91 Max.—159 Avg.—35.97	Min.—0.51 Max.—53.5 Avg.—21.69	Min.—3.92 Max.—59.00 Avg.—15.72
Zinc	μg/L	Avg.—37.00	Avg.—69.69	Avg.—BDL	Avg.—BDL	Avg.—BDL
Iron	μg/L	Avg.—3.73	Avg.—0.56	Avg.—0.35	Avg.—BDL	Avg.—0.19
Fluoride	μg/L	Min.—0.176 Max.—0.446 Avg.—0.315	Min.—0.137 Max.—0.282 Avg.—0.210	Min.—0.171 Max.—0.476 Avg.—0.254	Min.—0.162 Max.—0.393 Avg.—0.281	Min.—0.23 Max.—0.37 Avg.—0.27

N.B.: Min.—Minimum, Max.—Maximum, Avg.—Average, BDL—below detectable limit.

3.15 Pollution Indicator Species

The presence of Coliform bacteria and algae like *Navicula sp, Nitzschia sp., Oscillatoria sp.* acts as bioindicators of water pollution. This report was stated by Geelani et al.(2012). Jain et al.(2010) evidenced that zooplankton like *Alona guttata, Mesocyclops edax* are zone based indicator of water pollution whereas *Brachionus angularis and Keratella tropica* are also performed as trophic status and high-turbidity indicators, respectively, (Chicote et al. 2018). Besides, the changes within the plankton community represent the state to establish the trophic condition of the aquatic environment (Pradhan et al. 2008).

4 Conclusions

The negative effect on ecological and socio-economic view is well known to us. To protect the aquatic ecosystem of the Kangsabati river mass awareness, regular monitoring, unique planning, applying guidelines, and approaches for the management of human activities must be executed by several academicians, social workers, scientific organization, and concerned Government Department. Integrated management of river health and water resources under Department of Water Resources, River Management, and Ganges Rejuvenation, Central Water Commission, Government of India should be taken part actively to achieve the desired goal.

To mitigate the stress as well as anthropogenic effect a number of simple and trouble-free methods will be introduced for saving the health of the Kangsabati River based on nature and lives for future reference. Conservative dredging of sediment, quarrying of sand maintaining the balance of river bed, and plot-wise collection of soil for brick materials to recuperate the associated low land is to be firmly accomplished. The introduction of dumping ground for the purpose of recycling of solid wastes can rectify the risk and do well. To regulate the extreme river flow during monsoon and to avoid flood effect necessary steps should be taken promptly for long-term viability of the riverine ecosystem.

4.1 River Conservation Efforts

The government, the scientific society, and also the public have been aware of the difficulty of water pollution, reducing fish production, escalating incidence of floods, and scarcity of water resources in the Kangsabati River. The National Water Policy Ministry of Water Resources, Government of India (1987) emphasizes on improvement, utilization, management, and up keeping of water resources based on human needs like drinking water giving topmost priority, irrigation, hydropower generation, transportation, industrial use, and further operations. So, to conserve the water resources for future reference the following recommendations are very much urgent for implementation.

4.2 Recommendations

1. Human activities that cause pollution must be controlled creating consciousness.
2. Different regulations and guidelines should be implemented by industries or commercial site close to river to minimize or eradicate pollution.
3. Regarding the ecological significance of rivers, conservation measures like forestation, slope stabilization need to be taken.
4. A scientific management plan in relation to agricultural practices, urban expansion, waste management is urgently needed to further curtail the degradation of Kangsabati River Health.
5. Development of policies and eco-friendly methods should be introduced.
6. Long-term monitoring on-site of State Government as well as River Conservation Authority should be implemented.
7. Industrial effluents and municipal waste should be treated and recycled.
8. It is urgently needed for continuous examination and supervision of physico-chemical, biological, and bacteriological parameters of concerned river water taking appropriate remedial actions for the maintenance of water quality.
9. The conservation and sustainable management of the aquatic ecosystem of the Kangsabati river due to anthropogenic activities requires an Ecosystem Approach involving the restoration of the health of the water body coupling with the management of concerned biodiversity.
10. River health checkup is to be followed regularly.
11. Reutilization and recycling of waste—industrial effluents, sewage, and municipal waste can be recycled. For removal of pollutants—chemical, biological or heavy metals, the physicochemical techniques are adsorption, ion exchange, electrodialysis, and reverse osmosis, etc.
12. For domestic sewage percolation tank to be structured.
13. Cattle washing should be avoided.
14. The dead bodies of animals throwing into the river, dead bodies of human combustion at riverside, and fecal matter discharge should be strictly banned.
15. Redbrick manufacturing factory by the side of the river—Mohanpur must be closed to protect productive soils, river ecosystem, and minimizing air contamination.
16. An inclusive management plan which addresses the multiple sectors (silviculture, irrigation, tourism, infrastructure development, urban expansion, waste management, etc.) is essential to further curtail the degradation of the environment. Looking into the cultural and ecological significance of the rivers, conservation measures like forestation, slope stabilization needs to be taken.

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Chapter 4
Role of Modern Biotechnology in the Era of River Water Pollution

Ajaya Kumar Rout, Sangita Dixit, Sujata Dey, Pranaya Kumar Parida, Manojit Bhattacharya, Sukanta Kumar Pradhan, and Bijay Kumar Behera

Abstract The aquatic environment is the governance system for sustenance of life on earth. Nowadays, aquatic bodies are under serious threat throughout the globe due to industrialization and other anthropogenic activities. Pollution has been a stress for aquatic organisms, plants, and humans along with the pressure from changing climatic conditions. Freshwater is only 2.5% of the total water available on the earth and out of total freshwater 1.2 is only available on the surface rest are either in glacial form or in groundwater. The protection of high-quality freshwater is essential to maintain a healthy riverine ecosystem. This is an approach to implement advanced scientific technologies such as; microbiology, molecular biology, biotechnology, bioinformatics, metagenomics and other omics platforms to observe, analyze, and manage the information obtained from aquatic ecosystems to develop superior biotechnological methods to maintain the quality of freshwater.

Keywords Aquatic ecosystem health · River pollution · Biotechnology · Omics approach · Pollution monitoring

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1 Introduction

Water is necessary for sustenance of life on earth and river is the major source of it. Freshwater is only 2.5% of the total water available on the earth and out of total freshwater 1.2 is only available on the surface rest is either in glacial form or in groundwater. India is bestowed with 29,000 km length of river network which not only provides water for a human beings for their daily requirement but also provides, “fish” as a source of animal nutrition, aquatic plants along with water for irrigation to grow various crops and vegetables. Rivers play a significant role in Indian tradition and culture. They are the lifeblood of the majority of people in towns, cities, villages and many of them are considered sacred, with people taking holy baths in rivers during festivals. Bathing, navigation, public supply, drinking, fishing, and irrigation are only a few of the water uses along each river stretch. At the same time, it serves as a receptacle for the discharge of urban sewage, industrial effluent, and solid waste dumping. River Pollution is addition of something which changes the natural quality or substances which make the water unfavorable for life in river water (Wisdom 1966). River pollution is having a long history and presently, it is exploited to a higher degree. During the eighteenth century, many industries came to Europe and pollution growth was also there. All the waste from the industries and the urban area is discharged into rivers either without treatment or with partial treatment. Many acts came into force during the same time to save the river water and fisheries from pollutants from different sources. Gas Works clause Act 1847, Salmon Fishing act 1861 and 1865; Setting up of Royal commission on the prevention of river pollution, 1865, River pollution act 1876, 1890; Royal commission on Sewage disposal, 1898 but no act was passed in parliament; 1923, Salmon fishing act extended to trout are few of the acts to protect water resources and aquatic organisms. The Water (Prevention and Control of Pollution) Act of 1974 aims to protect the water quality of different water bodies that have been identified as best uses.

Now a days, industries are the major problem for environmental pollution. It releases a wide range of harmful substances, solid wastes, microbes, and gases of different types into our immediate ecosystems, such as water, air, and soil. Water pollution has become a global problem, with developing countries bearing a disproportionately high burden as a result of their growth imperatives (Inyinbor et al. 2016; Rana et al. 2017). Pollution of our water sources is a threat to humans and the aquatic ecosystem since population growth has accelerated climatic changes (Palmate et al. 2017). Biotechnology provides a wide range of opportunities for addressing river pollution issues effectively. Among different biotechnological tools, the metagenomic approach of identifying the helpful microbiome in river sediment is one of the approaches for pollution management.

2 Metagenomics Approach for Detection of Water Pollution

Over the last few decades, there have been many instances of water pollution/contamination caused by accidents or illegal activities all over the world. For example, the Tuojiang River pollution incident occurred in 2014, the Songhua River benzene pollution incident occurred in 2005, the Colorado River pollution incident occurred in 2015, and so on. The water contamination was caused by a massive amount of chemicals spilling into the atmosphere, resulting in a drastic increase in contaminants, which could have a variety of effects on the health of water bodies. Viral, bacterial, and fungal pathogens, as well as protozoal hosts for extracellular DNA and intracellular pathogens, were among the pollutants (e.g., antibiotic resistance genes) (Turnbaugh et al. 2009; Qin et al. 2010). It is hard to culture these pathogens in routine monitoring because most pathogens are fastidious, slow-growing. Antibiotic resistance marker genes or targeted pathogens that hybridize primers or probes engineered can be determined using culture-based molecular methods such as qPCR (quantitative PCR). This conventional approach relies heavily on data monitoring, which is labor intensive.

The metagenomics (DNA/16S rRNA based) method is suitable for monitoring of microbial target (i.e., virus, bacteria, and protozoa) from polluted river water. We came to this conclusion after reviewing the following evidence: (1) advances technologies for metagenome sequencing and (2) the accessibility and use of computational tools for metagenomic data analysis. However, it would need to further develop to the quantitative measurements of samples, good accuracy and high resolution of functional classification and phylogenetic analysis result in a mixed water population to accurately meet the requirements of refined the monitoring of water quality (Turnbaugh et al. 2009).

3 NGS Platform for Metagenome Sequencing

Several next generation sequencing (NGS) platforms were used for complete metagenome sequencing, such as Ion Torrent, 454 pyrosequencing, Illumina, Nanopore, and PacBio (Table 4.1). Longer reads (2900 bp) can be generated by PacBio compare to the Illumina reading (50–300 bp). The generated read depends upon the quality as well as size of the DNA template. The key distinguishing attribute of these sequencing technologies is to generate a huge number of short reads length sequencing (typically 100–300 bp per reading) per run (Aird et al. 2011; Brandariz-Fontes et al. 2015). Also, shotgun sequencing library preparation required approximately 400 bp DNA fragments, so no need for high quantity DNA. However, producing reads with lengths shorter than the standard too fragmenting DNA would also affect the efficiency of sequencing. Therefore, protocol optimization is very important to reduce the related error levels and lapses in sequencing quality.

Table 4.1 Comparative account of NGS approaches

Sequencing technology	Read-length	Accuracy	Read per run	GB per run	Time per run
Sanger sequencing	400–900 bp	99.9%	NA	NA	20 min–3 h
Pyrosequencing (454GSFLX)	400 bp	99.9%	1 million	0.4–0.6	24 h
SOLiD	50 bp	99%	1.2–1.4 billion	25–60	1–2 week
Illumina	50–300 bp	98%	3 billion	22–50	1–10 day
Ion torrent	Up to 400 bp	98%	5 billion	100 MB–200 GB	2 hours
Pac-bio	2900 bp	90%	35,000–75,000	2 hr	30 min–2 h

4 Bioinformatics Tools Used for Sequence Analysis

For the metagenome 16S rRNA/DNA sequence analysis, several bioinformatics tools such as QIIME, DADA2, MG-RAST, UPARSE, MOTHUR, MetaStorm, and minimum entropy decomposition (MED) are used, depicted in Table 4.2. Because of the lack of library demultiplexing and taxonomy assignment, the QIIME tools aim to perform downstream analysis using existing high-throughput pyrosequencing methods. Using raw sequencing data, the PyCogent toolkit in QIIME is used to resolve the interpretation and database deposition issue (Cole et al. 2009; Caporaso et al. 2010). It also provides functionalities visualization that was necessary for high-profile analysis, including gene/protein network analyses, sample diversity statistical analysis of within- or between-sample, and graphical representation of the analyzed data (Cole et al. 2009). Whereas, MG-RAST is a web application open-access server that suggests the automatic functional and taxonomic analysis of metagenome sequences (Keegan et al. 2016). It uses three databases, i.e., SSU, Greengenes, and RDP as a reference database, and algorithms/tools such as adapter trim, DRISER, BLAT are used for sequence preparation and taxonomy classification. Similarly, UPARSE is a method for constructing operational taxonomic units (OTUs) from NGS reads that achieve high sequence recovery accuracy and enhances the richness of mock communities (Edgar 2013). OTUs were reconstructions of sequences representing recognized species in some samples and identical to a known biological sequence in others, while retaining a few chimaeras and unclassified sequences in others (Edgar 2013). MetaStorm (http://bench.cs.vt.edu/MetaStorm/) is a new open-access server that allows users to customize metagenomic data sets and performs statistical analysis (Arango-Argoty et al. 2016). Users should access their reference database to customize the metagenomics annotation and specific markers for taxonomic and functional gene annotation. MetaStorm has two main research pipelines: an assembly-based annotation pipeline and a standard pipeline for decoding annotations from existing web servers. Both of these pipelines can be selected by users individually or in groups. Finally, MetaStorm offers

Table 4.2 Comparison of 16S rRNA amplicon tools

Tools	QIIME2	MG-RAST	UPARSE	MetaStorm	MOTHUR	DADA2
Web address	https://qiime2.org/	https://www.mg-rast.org/	http://drive5.com/uparse/	http://bench.cs.vt.edu/MetaStorm/	http://www.mothur.org/wiki/Main_Page	http://benjjneb.githubio/dada2/index.html
Availability	Open source and command line	Online and open source	Open source, offline, and command line	Online and open source	Open source, source code and tutorial	Open-source software
Online link for research paper	https://www.ncbi.nlm.nih.gov/pmc/articles/PMC5956843/	https://bmcbioinformatics.biomedcentral.com/articles/10.1186/1471-2105-9-386	https://www.nature.com/articles/nmeth.2604	https://pubmed.ncbi.nlm.nih.gov/27632579/	http://www.ncbi.nlm.nih.gov/pubmed/19801464	http://www.ncbi.nlm.nih.gov/pubmed/27214047
Published data	2018	2017	2013	2016	2009	2016
Algorithm used	–	BLAT	UPARSE-OUT, UPARSE-REF	–	DOTUR, SONS, TreeClimber, LIBSHUFF, Đ-LIBSHUFF UniFrac	DADA, divisive amplicon denoising algorithm, Needleman–Wunsch algorithm
Feature annotation	Taxonomy annotation	Taxonomy and functional annotation	Taxonomy analysis	Taxonomy and functional annotation	16S rRNA gene sequence analysis	
Hardware requirement	Mac OS X, Linux or windows	Online	–	Online	Mac OS X, Linux, or windows	
Citation	623	2056	4700	26	13,446	2903

improved high-quality data visualization, so that researchers can test and modify taxonomy and practical description at various resolution levels (Arango-Argoty et al. 2016). Whereas MOTHUR is a complete single-piece software package for metagenome analysis and it allows to use to analyze community sequence data (Schloss et al. 2009). TreeClimber, SONS, DOTUR, LIBSHUFF, and UniFrac have all used MOTHUR algorithms in the past (Schloss et al. 2009). To boost the DADA algorithm, DADA2 is a new model for Illumina amplicon errors (Callahan et al. 2016). The result of DADA2 gives a sequence table and OTUs tables. It was determined how many times the ribosomal sequence number was revised in each sample (http://benjjneb.github.io/dada2/tutorial.html). DADA2 is a model-based technique for correcting amplicon errors without the need for OTU construction. It aims to find fine-scale variations in 454-sequenced amplicon data with as few false positives as possible (Callahan et al. 2016). However, the overall workflow for river metagenome is shown in Fig. 4.1.

Metagenome sequencing has been documented in some rivers such as Yamuna and Ganga (Mittal et al. 2019; Reddy and Dubey 2019; Samson et al. 2019; Behera et al. 2020b, a; Das et al. 2020), Apies River (Abia et al. 2018), Amazon River (Ghai et al. 2011), 11 Rivers of West and East of Norwich UK, (Reddington et al. 2020), Bighorn River (Hamner et al. 2019), Bogota River (Ruiz-Moreno et al. 2019), James River (Brown et al. 2015), etc. and observed that archaeal and bacterial communities across the confluence response to a variety of factors responsible for water pollution, including variations in temperature, climate change, the physicochemical parameter of water, agricultural runoff, and unregulated small-scale industry. The information regarding the common microbes is present in the polluted water as shown in Table 4.3. When combined with Meta-Transcriptomic data (RNA-Seq of

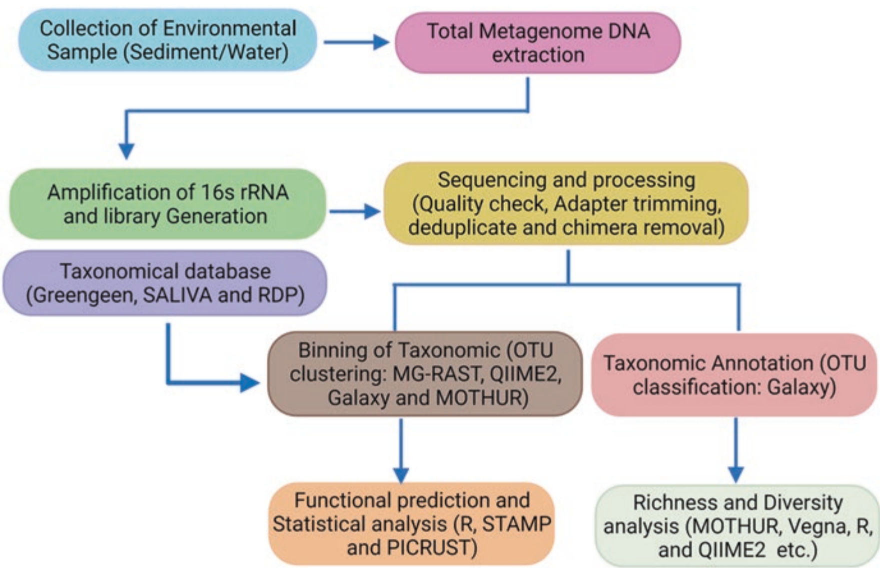

Fig. 4.1 Overall workflow for metagenome analysis

Table 4.3 A list of recent metagenome studies of river

Sl. No.	River	Sample type	Host microbes	References
1	Amazon	Water	*Actinobacteria, Proteobacteria*	Ghai et al. (2011)
2	Yamuna	Water	*Acinetobacter, Aeromonas*	Mittal et al. (2019)
3	Bogotá	Water	*Proteobacteria*	Ruiz-Moreno et al. (2019)
4	Ganga	Water and sediment	*Proteobacteria* and *Actinobacteria*	Reddy and Dubey (2019)
5	James	Water and sediment	*Proteobacteria, Acidovorax*	Brown et al. (2015)
6	Yamuna	Sediment	*Pseudomonas aeruginosa, Shigella sonnei, Escherichia coli, enterococcus faecium, vibrio cholerae, Streptococcus pneumoniae,* Acinetobacter baumannii, *Streptococcus suis, salmonella enterica*	Das et al. (2020)
7	Yamuna	Water	*Acinetobacter, Aeromonas*	Mittal et al. (2019)
8	Ganga	Sediment	*Streptomyces bikiniensis, Rhodococcus qingshengii,* Bacillus aerophilus, *pseudomonas veronii, Phanerochaete chrysosporium,* and *Rhizopus oryzae*	Behera et al. (2020a, b)
9	Yamuna	Sediment	*Lactobacillus* spp., *bacillus* spp.	Behera et al. (2020a, b)

environmental samples), metagenomics has shed light on group function and can provide information about the critical microbial survival in a given condition. The microbes identified from metagenomic analysis on biodegrading microbes will help in industrial use of river water pollution abetment.

5 Biofilms as Aquatic Environment Cleaning Technology

Biofilms are multifaceted permeable structures and one of the greatest tolerant communities belonging to the aquatic ecosystem. Ecologically efficient strains of bacteria, algae, and fungi contribute to its structural components. Aerobic microbes and algal components in biofilms free aquatic ecosystems from excessive nutrient loadings and pollutants by using them as carbon sources like potential degrading agents (Sgountzos et al. 2006). They can be utilized in the bioremediation of heavy metals and also as pollution bioindicators. Different studies have evaluated the role of biofilm-derived microorganisms to remove or reduce the effect of several contaminants from polluted water in an environment friendly way (Table 4.4). The applications are cost-effective and hold great future scopes. Microorganisms inside biofilms use the basic cellular machinery of assimilation, adsorption, and biodegradation to

Table 4.4 Biofilm-derived microorganisms to remove or reduce the effect of several contaminants from polluted water

Biofilm-derived microbial isolates	Environmental Benefits	References
Burkholderiae sp., *Sphingomonas* sp., *pseudomonas* sp., and *Chryseomonas* sp.	*Pseudomonas* MT1 isolate showed the highest biodegradation capacity (1700 mg/L of phenol and 3000 mg/L of pyridine)	El-Rakaiby et al. (2013)
Aeromonas caviae, *fusarium udum*, *pseudomonas stutzeri*, *Sphingobacterium thalpophilum*, and *Rhodotorula mucilaginosa*	Significant reduction in BOD and COD of wastewater	El Bestawy et al. (2014), Rozitis and Strade (2015)
E. coli (Rz6), Bacillus amyloliquefaciens (S1), and their mixed culture. *Pseudomonas otitidis*	Total suspended solids and total coliform from wastewater were eliminated. *Pseudomonas otitidis* has been able to degrade crude oil	Dasgupta et al. (2013), El Bestawy et al. (2014)
A. hydrophila L6, B. denitrificans B79, *Thauera* sp., *pseudomonas stutzeri* and *Rheinheimera pacifica*	Biological denitrification of wastewater and river water is enhanced by these microbes	Jiang et al. (2008), Andersson (2009), Sriu Naik and Pydi Setty (2011)
Azoarcus sp., *Thauera* sp., *Comamonas* sp. and *Paracoccus* sp.	Heterotrophic nitrification in aquatic ecosystems.	Wang et al. (2014), Ma et al. (2015), Cydzik-Kwiatkowska and Zielińska (2016)
Candidatus kuenenia and *Nitrosomonas* sp.	Excess ammonium was eliminated from waterbodies exhibiting high concentrations	Park et al. (2014)
Accumulibacter sp., *Tetrasphaera* sp. and *Dechloromonas* sp.	Phosphorus contamination was reduced from polluted aquatic environments	Kong et al. (2007), Oehmen et al. (2007), Nielsen et al. (2010), Nguyen et al. (2011)
Sphingomonas bisphenolicum, *pseudomonas paucimobilis*, *Sphingomonas* sp. AO1 and *P. mendocina* NR802	Micropollutants of aquatic bodies, such as polycyclic aromatic hydrocarbon (PAH) and bisphenol A, had undergone biodegradation	Mangwani et al. (2014), Cydzik-Kwiatkowska and Zielińska (2016)
Geobacter metallireducens and *pseudomonas putida*	Effective bioremediation of heavy metal contaminated water bodies	Singh and Cameotra (2004)
Fusarium sp., *Graphium* sp., *Candida tropicalis,* and *Acinetobacter* sp.	Removal of chemicals like phenol and m-cresol from wastewater	Santos and Linardi (2004), Ying et al. (2007)
Bacillus cereus AUMC B52 and *P. aeruginosa* ASU 6a	Acts as cost-effective bio-sorbent material for the elimination of Zn (II) from wastewater	Joo et al. (2010)
Pseudomonas maltophilia, Acinetobacter calcoaceticus, *P. aeruginosa* and *Erwinia herbicola*	Plays a significant part in the bio-sorption of metals like gold (au)	Tsuruta (2004)

(continued)

Table 4.4 (continued)

Biofilm-derived microbial isolates	Environmental Benefits	References
Escherichia coli	Potential bioremediation agent for several heavy metals, such as cadmium (cd), lead (Pb), zinc (Zn), and copper (cu)	Kao et al. (2006)
Sphingomonas sp. and *Sphingpoyxis* sp.	Effective against specific toxic compounds like aliphatic-aromatic copolyesters, microcystin-RR, and aliphatic homopolyesters	Abou-Zeid et al. (2004), Wu et al. (2010)
Trichoderma sp., A. niger and basidiomycetes	Decreases dye contamination of water (Congo red, Orange G.) by biosorption	Tatarko and Bumpus (1998), Sivasamy and Sundarabal (2011)

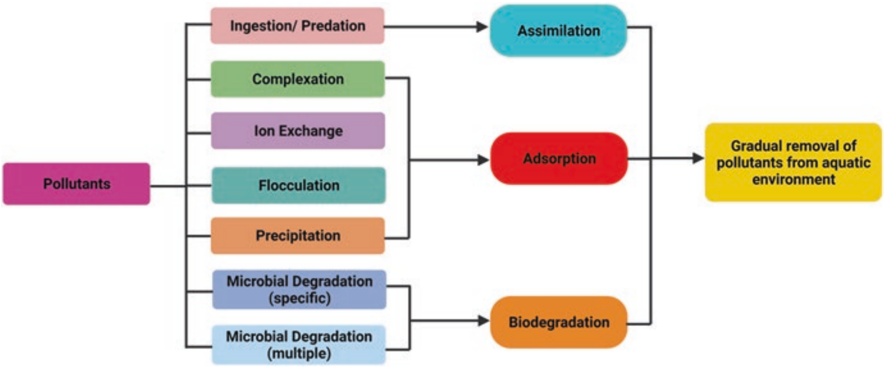

Fig. 4.2 Schematic representation of removal of pollutants from an aquatic ecosystem (Wu et al. 2012)

remove toxins from contaminated water. Figure 4.2 depicts the whole process of removing toxins from the aquatic environment (Wu et al. 2012).

6 Removal of Pesticide from Polluted Water through Remediation Technology

Remediation is one of those promising technologies that have the potential to reverse a contaminated area to pristine condition (Fig. 4.3). The economically feasible and ecologically effective techniques reduce the effect of chemical pollutants in contaminated sites. A combined approach of microbial and phytoremediation (Table 4.5) ensures a comparatively effective clean-up of polluted sites. Villaverde et al. reported *Advenella* sp., *Variovorax soli,* and *Anthrobacter sulfonivorans* of microbial consortium mineralized Diuron, an active herbicide or algicide found as

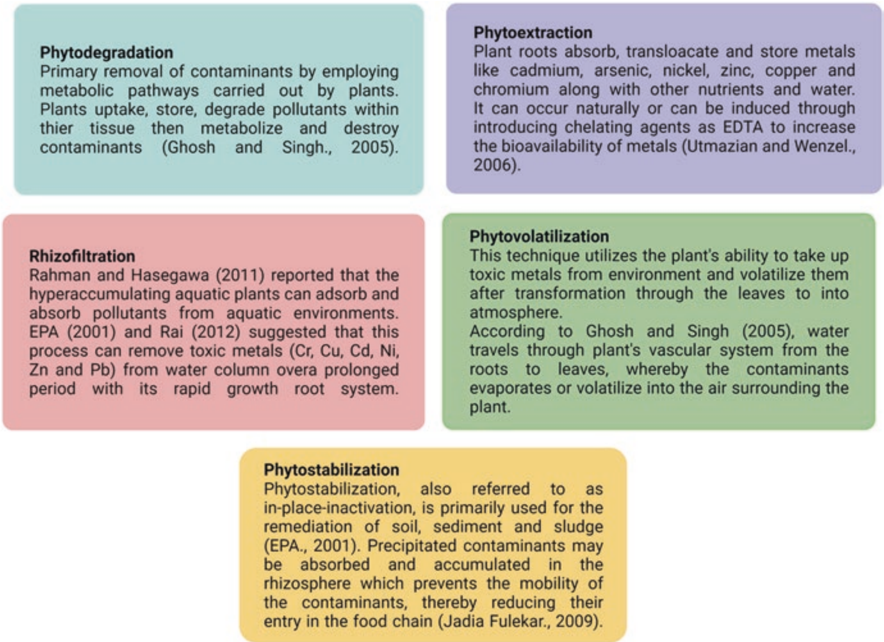

Fig. 4.3 Phyto management technologies to combat water pollution

an environmental contaminant (Villaverde et al. 2017). Briceno et al. observed that a mixed culture of Streptomyces sp. exhibited the high removal of diazinon pesticides (Briceño et al. 2016). Development of Mycelia and enhanced growth rate of *Streptomyces* sp. holds the remediation potential of pesticides (Fuentes et al. 2017). The microbial remediation technique is another effective tool that eliminates pesticides from polluted environments.

Several fungal intracellular enzymes, such as cytochrome oxygenase, reductase, and methyltransferase participate in the organic pollutant remediation. According to reports from various studies, fungal strains are involved in the remediation of pesticides, such as *Phanerochaete* sp. (Chirnside et al. 2011), *Aspergillus* sp. (Mohamed et al. 2011), *Penicillium* sp. (Peng et al. 2012), etc. The mycelial networks along with the capacity to utilize organic material as a source of nutrients make fungi a suitable bioremediation agent (Maqbool et al. 2016). *Phanerochaete sordida* (White rot fungus) employs extracellular enzymes to convert clothianidin (a pesticide) into a nontoxic metabolite, N-(2-chlorothiazol-5-methyl)-N-methyl urea (TZMU) (Mori et al. 2017).

Spirulina platensis hydrolyzes chlorpyrifos in presence of alkaline phosphatise to 3, 5, 6-trichloro-2-pyridinol which is a nontoxic primary metabolite (Thengodkar and Sivakami 2010). The microalgal species *Chlamydomonas mexicana* degrades atrazine (Kabra et al. 2014).

Cichorium intybus and Brassica juncea are reported as efficient plants in the degradation of DDT. According to Syuhaida et al. aquatic plants such as *Lemna minor*, *Elodea canadensis,* and *Eichhornia crassipes* exhibit high photosynthetic

Table 4.5 List of plant species used in pesticide remediation

Pesticide	Plant	References
Endosulfan	*Brassica campestris, Solanum lycopersicum*, *Medicago sativa Helianthus* sp. and *glycine max*	Mukherjee and Kumar (2012); Mitton et al. (2016)
Azoxystrobin	*Plantago major*	Rai et al. (2015)
Fenpropathrin	*Spirodela polyrhiza*	Xu et al. (2015)
DDT	*Medicago sativa, Glycine max*, *Solanum lycopersicum*, and *helianthus* sp.	Mitton et al. (2014)
Endosulfan sulfate	*Zea mays*	Somtrakoon et al. (2014)
Lindane	*Jatropha curcas* L.	Abhilash et al. (2013)
Cypermethrin	*Pennisetum pedicellatum*	Dubey and Fulekar (2013)
Hexachlorobenzene	*Typha latifolia*	Zhou et al. (2013))
Atrazine	Acorus calamus	Wang et al. (2012)
Trifluralin, Metalaxyl	*Salix alba*, *Sambucus nigra*	Fernandez et al. (2012)
Atrazine, Diazonin, permethrin	*Leersia oryzoides*	Moore and Locke (2012)
Phoxim	Allium fistulosum	Wang et al. (2011)
HCHs	*Withania somnifera*	Abhilash and Singh (2010)
Dimethoate, malathion	*Nasturtium officinale*	Al-Qurainy and Abdel-Megeed (2009)
Chlordane	*Cucumis sativus*	Gent et al. (2007)
DDE	*Brassica juncea*, *brassica napus*	White et al. (2005)
Aldicarb	*Zea mays*, *Vigna radiate*, *Vigna unguiculata*	Sun et al. (2004)
Butachlor	*Triticum vulgare*	Yu et al. (2003)
Chlordane	*Spinacia oleracea*	Mattina et al. (2003)

activity, high pollutant absorption rate, high growth rate, and easy harvesting, which is being used in water treatment (Syuhaida et al. 2014).

According to Chuluun et al. (2009), Organophosphate (malathion, parathion, diazinon, and fenitrothion) and Organochlorine (dieldrin, HCB) pesticides can be absorbed from the aquatic ecosystems by A. Gramineus (Chuluun et al. 2009). The potential of Atrazine removal of *Lemna gibba* and Azolla caroliniana has also been reported by (Guimarães et al. 2011). Macrophyte species *S. polyrhiza* and *L. minor* exhibited the highest rate of removal efficacy for the two particular fungicides, pyrimethanil, and dimethomorph (Dosnon-Olette et al. 2009).

7 Floating Treatment Wetlands as a Remediation Agent of Contaminated Water

A novel technology that offers great potential for the treatment of polluted aquatic ecosystems is Floating islands in Wetlands (Ladislas et al. 2015). Emergent wetland plants developing hydroponically on structures form floating treatment wetlands

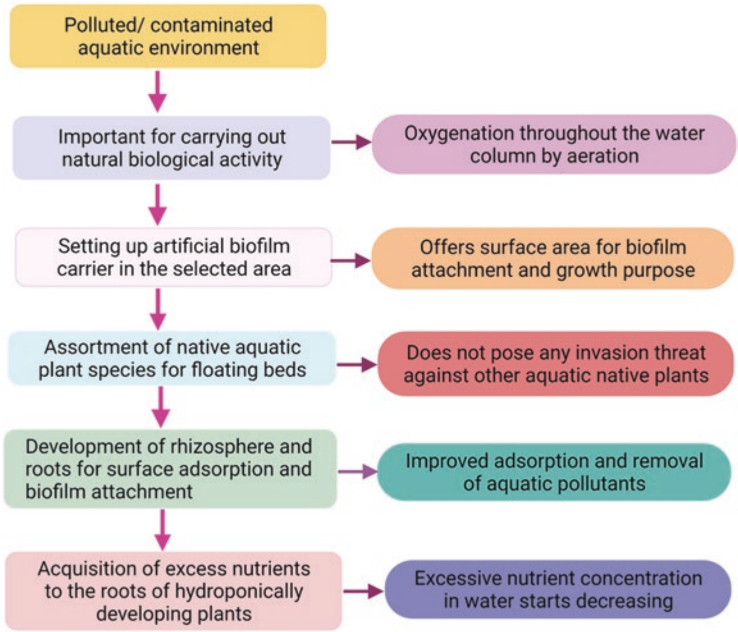

Fig. 4.4 Schematic representation of a floating treatment wetland (modified version of Samal et al. 2019)

(FTWs) (Fig. 4.4). The floating islands provide essential nutrients to the microbial community, whose metabolic by-products are utilized as resources for other organisms forming a complex network in a wetland (Chaudhuri et al. 2014). A large surface area developed by the plant roots enhances the capacity for adsorption, biofilm attachment, and storage of nutrients (Tanner and Headley 2011). FTWs are able to withstand profound and fluctuating water levels when they expand through the floating mat and into the water below. This approach can be regarded as a cost-effective alternative along with ecological benefits.

8 Conclusion

Bioremediation of environmental contaminants/pollutants has become a major challenge as public awareness is growing day by day on the harmful effects of pollutants on living organisms. Biotechnology is a rapidly growing field that employs microbial activity as a biological treatment for a variety of environmental issues. Metagenomic approach and other biotechnological approach have their own advantages and disadvantages. However, they give an idea of utilization of these technologies for a beneficial impact on the aquatic environment. The aim of “The Water Act

1974" should be to conserve and restore river water's wholesomeness in terms of ecological sustainability. The biotechnological tools will help in achieving this objective.

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Chapter 5
Assessment of Chitinolytic Bacteria Isolated from Zooplankton of Freshwater Ecosystem

Sujoy Midya and Sk Saruk Islam

Abstract Rivers represent one of the major freshwater sources of the aquatic ecosystem in our planet. Riverine health is of utmost importance to determine the health condition of the aquatic ecosystem. Nowadays, urbanization and anthropogenic impacts convert freshwater sources to polluted zones through the deposition of different waste materials. Urban waste composition makes serious damage to the physicochemical properties of the freshwater system that can indirectly hamper the nutrient cycle of the aquatic system. This mutant environment can facilitate the growth of different opportunistic organisms like microbes. Chitinase-producing bacteria in freshwater ecosystem contribute not only to nutrient cycling and energy flow by causing degradation and utilization of chitin but also develop a mutualistic association with zooplankton by getting shelter on the one hand and nutrients after digesting the chitin of the zooplankton on the other. In the present study, a total of thirty-five bacterial isolates were screened from zooplankton *Heliodiaptomusviduus*, based on clear zone/colony size ratio of which eight isolates based on their growth in colloidal chitin agar media have been identified as to be potential chitinolytic ones. These isolates have also been observed to exhibit chitinolytic activity in respect of CZ/CS ratio within a range of 1.41–1.75. The characterization of those bacterial strains possessing the power of higher chitinase production was performed by biochemically and enzymatically characterized Vitek2 compact automated system, following the subsequent confirmation by16 s rRNA sequencing. Two novel bacterial strains, namely, Aeromonas sobria and Enterobacter aerogenes, were identified in these sequencing protocols. The enzymes designated as β-N-acetyl-glucosaminidase and chitinase have been assessed, and maximum enzymatic activities were demonstrated by *A. sobria* (161.7 Uml-1 and 18.02 Uml-1, respectively) and Enterobacter aerogenes (141.8 Uml-1 and 15.78 Uml-1, respectively). The present research findings have unearthed some new and interesting research

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information pertaining to the trophic interactions through the exchange of biochemical entities in an aquatic ecosystem, the baseline knowledge of which are expected to be utilized in the ecosystem management from the point of view of productivity and maintaining of the ecosystem health.

Keywords Aquatic ecosystem · Biochemical monitoring · Ecosystem health · Isoenzyme sensitivity · Vitek 2 compact system

1 Introduction

The aquatic ecosystem represents a habitat for aquatic life. This unique habitat also acts as a pool of potential persistent different chemicals of the environment. The innovation of technological advancement, the increasing amount of urban waste materials, and a rising density of population have often resulted in the conversion of aquatic zones like lakes, ponds, rivers, and coastal zones into waste depots, where the natural equilibrium is severely concerned (Sharma 2018). Some of the heavy metal pollutants like arsenic (As) and cadmium (Cd) have drawn more attention for human life. They are creating global concern due to their possible toxicity, the impact of severity, persistence in the environment, and ability to bioaccumulate through the living component in aquatic ecosystems (Batvari et al. 2015). The major sources of contaminants that are introduced into the aquatic system are from various sources, either naturally occurring deposits or through anthropogenic activities. Heavy metals are trapped naturally in the lentic and lotic environment and accumulated in sediment, being directly available to benthic organisms. Now heavy metals are released to the water column through different ways that create the necessity dissolved concentration of oxygen in the aquatic environment which threatens the aquatic ecosystem (Pekey et al. 2004). In addition to sediment and water, these elements can also enter the food web through prey–predatory relationship (zooplankton, phytoplankton, and benthos) or by uptake through the gills and skin of aquatic organisms and be potentially accumulated in fish in aquatic ecosystems (Ahmed et al. 2005). This mobilization makes that they can be accumulated in the tissues of the body of the living organisms (bioaccumulation) and transferred through the aquatic food chain, increasing the concentration of the heavy metal as they pass from lower to higher trophic levels of the food chain (biomagnification). Heavy metals accumulation in the tissue of the species depends on a number of factors like toxicity, physicochemical conditions, and exposure routes (Croteau et al. 2005). Both the biological and the ecological factors have also influenced the mechanism of the detoxification of species (Soto-Jiménez 2011). Other studies have explained the behavior and relationships of heavy metals in the biotic and abiotic components of the aquatic ecosystem (Aderinola et al. 2009). Most of the research on these pollutants has focused on isolated components like sediments (Aprile and Bouvy 2008), water (Melgar et al. 2008), and plants (Bayen 2012). Bioaccumulation and biomagnification of these heavy metals pollutants (inorganic elements) in all types of components of an aquatic ecosystem are elevated due to pollution. Hence, the understanding of the dynamics of pollutants in the aquatic environments is very essential.

In the aquatic ecosystem, microorganisms exhibit different forms of metabolisms through the intricate pathways of trophic interactions. A few microorganisms are phototrophic (primary producers) referred to as phytoplankton. They are recognized as the aquatic microscopic plants that use dissolved nutrients from the surrounding environment, inorganic carbon, and solar energy to make their own biomass, and phytoplankton produces and releases organic products, referred as autochthonous dissolved organic carbon (DOC), which make up major food sources for higher trophic levels of this aquatic environment (Reynolds 2008). The autochthonous DOC is now accessible to be consumed by bacteria that are heterotrophic in nature, or referred as secondary producers, which are also recognized as bacterioplankton. Bacterioplankton is small and prokaryotic. They are consuming dissolved organic for obtaining energy that helping for their growth and reproduction (Kirchman et al. 2008). The bacterioplankton is using DOC as a food source and then is being harvested by bacterivorous organisms (unicellular). Heterotrophic flagellates and ciliates are considered bacterivorous organisms (Kirchman et al. 2008). Both phytoplankton and the unicellular bacterivorous organisms are used as a food source for multicellular zooplanktons (Rolff and Elmgren 2000; Berglund et al. 2007). Then, small fishes predate multicellular zooplanktons that link-up between the microscopic biosphere and macroscopic organisms, and it helps to supply nutrients and energy to larger organisms of the aquatic ecosystem (Karus et al. 2014). Both the trophic levels, that is, phytoplankton and bacterioplankton, form the baseline of the food web in aquatic ecosystems by using pray–predatory relationship (Berglund et al. 2007).

Chitin, representing a prime structural chemical constituent of the cuticle of crustacean copepods, is characterized in having some unique physical and chemical properties, such as insolubility in water, even in diluted acids and alkalis, and the complex meta-porphyrin ring is constituted by linear chains of β-1,4-linked homopolymer of N-acetylglucosamine (GlcNAc)(Wendisch et al. 2016; Yan et al. 2017). This complex polysaccharide ranks second in nature after cellulose as per their availability, amounting to an annual production of about 100 billion tons (Yu et al. 1993). Chitin is a common structural constituent of the cell wall of fungi, exoskeletons, and shells of crustaceans. The enzyme chitinases (EC 3.2.1.14) are the glycoside hydrolases (GHs) that degraded chitin by hydrolyzing β-1, 4-glycosidic linkages (Jarmila and Vavrikova 2011). Based on amino acid sequencing of the catalytic domain, chitinases are classified into two different families of glycoside hydrolases, and these families are marked by numbers 18 and 19 (Henrissat and Bairoch 1993; Varum and Smidsrod 2005). Chitinases under the family 18 are found in a wide range of organisms, including microbes, plants, and animals, whereas chitinases of family 19 are predominantly found in plants and relatively rare groups of prokaryotic organisms in nature (García-Fraga et al. 2015). The chitin-producing genes after their isolation from various chitinolytic bacteria have been cloned, analyzed, and their biochemical properties have also been recorded (Hashimoto et al.

2000; Prashanth and Tharanathan 2007; Bhatnagar and Sillanpa 2009). A good number of studies have reported the roles played by the chitinases and chitinolytic bacteria in inhibiting the mycelial extension of various types of pathogenic fungi (Itoh et al. 2006; Ghormade et al. 2017), which directly substantiated the active and critical roles played by bacterial chitinases in the digestion of chitin present in the fungal cell walls and could be widely useful as environmentally friendly for future agents for the biocontrol of agricultural phytopathogens (Kamensky et al. 2003). Several numbers of chitinolytic bacteria and their chitinases activity that have been studied against phytopathogenic control have been isolated from different sources like soil and marine environments while infrequent from freshwater environments (Rahman et al. 2016). Thus, analysis of chitinolytic bacteria and characterization of their chitinases activity from the freshwater environments are crucial for better understanding of their function and efficiency against phytopathogen in agriculture. A biofilm is formed by single or multiple species of microorganisms (Kjelleberg and Givskov 2007; Seneviratne et al. 2008). And it has been shown to protect microorganisms from different types of stresses of the environment (Singh et al. 2006). Chitinolytic bacteria that form biofilms can be firmly attached through the mycelia of fungi. Chitin in cell walls of fungi can induce chitinase expression in chitinolytic bacteria, and the production of chitinases has been degrading such chitin as alternative source of nutrition, resulting in fungal destruction (Meziane et al. 2006; Hoster et al. 2005; Xia et al. 2011). Considering both the common and the large abundances of crustacean zooplankton in aquatic environment environments, the interaction between zooplankton and bacteria can widely affect biogeochemical activities (Dattagupta et al. 2009). In times of rapid climate change, there is an urgent need for a proper understanding of both zooplankton–bacteria interactions and how they would react to future climate scenarios. The approach used in this study explains the screening of freshwater bacteria for those that possess high chitinase activity and generate biofilms from freshwater zooplankton (*Heliodiaptomusviduus*, Gurney 1916), and we examined the chitinase activity of the selected bacteria and the culture supernatant.

2 Materials and Methods

2.1 Study Area

Based on topography, land use patterns, sources of run-off load, and human impact, five subsites of the freshwater sources of Kangsabati river having contrasting ecological factors were selected from the Midnapore (West) district of West Bengal, India. Water samples containing suitable moderate size species of copepod zooplankton (*Heliodiaptomusviduus*, Gurney 1916) were collected, acclimatized in the laboratory condition, and cultured with the collected natural water maintaining the pH (7.2–7.8) and temperature (27–30 °C) from water body of freshwater of Shal

Dahari river bank (22° 29′ 20.15′ N 87° 33′ 29.59′ E), Pathra river bank (22° 24′ 27.00′ N 87° 25′ 1.67′ E), Najargunj river bank (22° 24′ 3.73′ N 87° 18′ 45.15′ E), Payraguri river bank (22° 27′ 39.86″ N 87° 5′ 27.83′ E), and Balishira river bank (22° 29′ 44.60′ N 87° 4′ 6.24′ E), Midnapore (West) District West Bengal, India. These sites were situated in comparatively low-lying areas having the predominance of natural vegetation surrounding the study site within the closest vicinity of the cultivated areas.

2.2 Plankton Density and Water Physiochemical Property

Following the formula, that is, Individuals/L = {A* (n/v)}/ L, was used to calculate the number of plankton species present in the Sedgewick rafter cell. The number of cells per mm was multiplied by a correction factor to adjust the number of organisms per litter (Dussart and Defaye 2001a). Zooplankton abundance has been expressed as a number of individuals per litter of the sample. Moreover, wetlands water parameters were measured in situ using the Water Analyzer (TOWA, Japan, Model No- WQC-22A). Water transparency was measured with a 20-cm diameter Secchi disk.

2.3 Isolation and Screening

The identification of zooplankton (copepod) was done to the lowest taxonomic level following standard literature and with the help of phase-contrast microscope (Nikon E200) (Battish 1992; Dussart and Defaye 2001b; Midya et al. 2018). Out of the collected species, the most abundant copepod, *Heliodiaptomusviduus*, under the order copepod, class crustacean, and phylum Arthropoda (Gurney 1916), was selected for the detailed study. The collected zooplanktonic copepods were homogenized and incubated in Alkaline Peptone Water (APW) at 37 °C for a period of 18–24 h for bacterial growth (Midya et al. 2019).

2.4 Colloidal Chitin Agar (CCA) Preparation and Screening of Chitinolytic Bacteria

Flake chitin (HiMedia, India) with an amount of 5 g after mixing with 90 mL ice-cold HCl (10 N) was kept overnight in shaking condition. Then, the suspension was mixed with the ice-cold ethanol (50% v/v) and placed at 25 °C for overnight with rapid stirring. The precipitate of the sample was collected by centrifugation (10,000 *g* for 10 min) and washed with sterile distilled water several times to bring the pH to a neutral level (Halder et al. 2012; Korany et al. 2019). Chitinase-producing bacteria were isolated from the subsurface of APW medium on colloidal chitin agar media (CCA) (Media composition (w/v): colloidal chitin, 2.0%; peptone 0.25%; beef extract 0.25%; agar 2.0%) by dilution plating technique. After 72 h of incubation at 30 °C, the chitinolytic bacterial colonies were examined based on their appearance of a clear zone of chitinolysis around the colony, and the potent strains were screened based on the chitinolytic index (Bradner et al. 1999).

2.5 Biochemical Characterization and Assessment of Enzyme Sensitivity through Vitek2 Compact System

To detect biochemical responses, cultures were streaked on a plate count agar media, and thereafter, single colonies were screening and identified by biochemically using Vitek2 compact (BioMérieux), an automated system determining the microbial growth potentiality colorimetrically (David 2009). Colonies isolated from pure culture medium were allowed to experience suspending in 3.0 mL of normal saline water (0.50% NaCl, pH- 6.5), maintaining the turbidity to range between 1.80 and 2.20 (McFarland). The microbial identification cards were suspended in microbial suspension shadowed by card sealing, during incubation at 36 °C temperature, and the interpretations were taken accordingly. Bacterial enzyme properties were also ascertained by an automated Vitek2 compact system (bioMérieux, India) which could integrate various isozymes of aryl amidase, galactosidase, glucosaminidase, glucosidase, transferase, and xyloasidase (Ganguly and Chakraborty 2018). Positive data as derived from the Vitek-2 compact system for these bacteria with respect to the specific set of nitrogen and carbon turnover enzymes worked as a potential contributor.

2.6 Assay of β-N-Acetyl-Glucosaminidase and Chitinase Activity

The microbial broth culture was centrifugated at 5000 rpm for 5 min, and the supernatant was considered as crude enzyme. To perform assaying of *β*-N-acetyl-glucosaminidase (chitobiose; EC 3.2.1.30) activity, the crude enzyme was measured according to Roberts and Selitrennikoff (1988) with some modifications (Aktuganov et al. 2019). The reaction mixture was prepared with 50 μL of enzyme solution which was mixed with 100 μL of 150 m M*p*-nitrophenyl N-acetyl–β-d-glucosaminide (HiMedia, India) and 200 μL of 0.1 M phosphate buffer (pH 7.0). After incubation at 30 °C for 15 min, the reaction was stopped with the addition of 50 μL of 1 N sodium hydroxide. To assay chitinase (EC 3.2.1.14) activity, the crude enzyme was measured following Schale's method (Imoto and Yagishita 1971) with some modifications (Dinh et al. 2018; Berini et al. 2016). In all, 400 μL reaction mixture contained, 0.1% of 100 μL colloidal chitin as substrate and was mixed with 50 μL enzyme solution and 200 μL of 20 mM sodium phosphate buffer (pH 7.0). After incubation at 30 °C for 15 minutes, the reaction was stopped with the addition of 50 μL of 1 N sodium hydroxide. The amount of released *p*-nitrophenol and N-acetyl glucosamine was measured spectrophotometrically at 405 nm in UV–VIS spectrophotometer (Techcomp, Hong Kong). One unit (U) of chitinase activity was defined as the amount of enzyme needed to release 1 μmol of p-nitrophenol and N-acetyl glucosamine per minute under standard enzyme assay conditions.

2.7 16 S rRNA Sequencing and Phylogenetic Tree Construction

The 16 s rRNA sequencing was performed to identify the bacterial samples after their isolation from the samples of copepods. Bacterial genomic DNA extraction was done through the heat shock method, which involved three heat shock cycles, each with two environmental conditions (20 mins at −85 °C temperature and 15 mins at 95 °C temperature). Amplification of genomic DNA samples was carried out by two universal eubacterial primers 27F (5-AGAGTTTGATCMTGGCTCAG-3) and 1492R (5 -AGAGCCCGATCMTGGCTCAG-3) using PCR. In all, 25 μ1 master mix contained template DNA-2 μ1 (100 ng), enzyme: Taq polymerase-0.5 μ1 (3 U/μL) (Genei), 1.5 μ1 of 10x Taq polymerase buffer contains (100 mM Tris (pH 9), 500 mM KCl,15 mM MgCl2, 0.1% Gelatin) (Genei), 1.5 μ1 of dNTP mix (10 mM) (Genei), 1 μL each primers (5pM/μL) (Eurofin), and rest volume made up by molecular-grade water. The reaction conditions were 94 °C for 5 min, 94 °C for 45 s, 56 °C for 1 min, 72 °C for 1 min (for 35 cycles), and then 72 °C for 10 min and final holding at 4 °C. Sequencing was done by outsourcing from Scigenom, Kochi. Using the NCBI BLAST (www.ncbi.nlm.nih.gov) search engine, the gene sequences were compared with others in the GenBank databases. Best-quality sequences of representative organisms were obtained from databases and aligned with our isolated sequence (Veronica and Kalisa 2019). Chromatograms of sequences were edited and analyzed by Bio-edit 7.0.9.0 multiple sequence alignment program, for phylogenetic analysis (Flavioet al. 2017). Finally, a phylogenetic tree was constructed with the aligned sequences in MEGA7.0 program using the neighbor-joining method with 1000 bootstrap replications (Flavio et al. 2017; Holmes 2003).

2.8 Statistical Analysis

All assays for estimating activities of *β*-N-acetyl-glucosaminidase and chitinase activity assay were carried out in triplicate, and the results were statistically processed to derive the values of means with a standard deduction (±SD). Data obtained from the studies of enzymatic activities were statistically analyzed using one-way ANOVA in IBM SPSS version 25. Significant differences in enzyme activities were determined based on Tukey's test at $P < 0.05$ (Adams and Collyer 2018).

3 Results

3.1 Abiotic and Plankton Density

The water physiochemical variables of different selected study sites showed a marked variation in the study sites. In these studies, surface water temperature varied from 28.15 ± 3.21 to 29.67 ± 4.03 °C in the study sites. Dissolved oxygen concentrations were between 3.94 ± 0.47 mg/L and 5.12 ± 0.53 mg/L at the sites. The highest salinity showed at study site-V (SV) 16.16 ± 3.78 ppt, whereas lowest salinity was found in study site-I (SI)13.73 ± 3.05 ppt. Secchi depth, which was used as an indication of water transparency, was varied from 47.68 ± 5.83 cm to 49.75 ± 6.61 cm throughout the whole period in study sites SII and SI, respectively. The average pH varied in a range of 7.21 ± 0.39 to 7.80 ± 0.86 in the selected study area. Plankton density was also highest in study site-II (SII) 1256 ± 26.52Ind/L. Other water physiochemical factors are shown in Table 5.1.

3.2 Microbiological Study

In the present study, the total bacterial counts (TB) and chitinolytic bacterial isolates (CB) were screened from different specific mediums (Fig. 5.1). New bacterial isolates tended to exhibit their growth in high colony-forming unit (cfu). The maximum density of total bacteria and chitinolytic bacteria were found to be 7.71×10^8 cfu/g and 3.37×10^4 cfu/g, respectively, at study site-II (SII), while the chitinolytic bacterial counts were lower than the total bacterial counts. On the other hand, study site-IV (SIV) showed the lowest total bacterial (6.21×10^8 cfu/g) population, whereas study sites-V (SV) has the lowest chitinolytic bacterial (1.67×10^4cfu/g) population. Total bacterial populations are much higher than normal bacterial counts as obtained in most of the microbial studies (above 10^7cfu/g) which indicated ecologically disturbed or risky health of the ecosystem (Bomar 1988). So, the present study revealed that both bacterial populations excised the health risk indication. Thirty-five (35) chitin-degrading bacteria were isolated from the samples of crustacean zooplankton copepods of which eight (8) bacterial isolates were selected from specific medium depending on their higher chitinolytic zone/colony size (CZ/CS) ratio (Fig. 5.2), which were further subjected to different microbiological parameter analysis. Previous researchers established that

Table 5.1 Water physiochemical parameters of five selected study sits and plankton density

Water parameter	Unit	Study sites				
		SI	SII	SIII	SIV	SV
Temperature	°C	28.86 ± 4.09	28.67 ± 2.94	28.15 ± 3.21	29.67 ± 4.03	28.92 ± 3.68
pH	pH scale	7.45 ± 0.68	7.21 ± 0.39	7.58 ± 0.44	7.69 ± 0.58	7.80 ± 0.86
Dissolve oxygen	Ppm	5.12 ± 0.53	3.94 ± 0.47	4.40 ± 0.39	4.92 ± 0.85	4.67 ± 0.82
Salinity	Ppt	13.73 ± 3.05	14.64 ± 2.84	15.70 ± 3.13	14.79 ± 3.71	16.16 ± 3.78
Secchi depth	Cm	49.75 ± 6.61	47.68 ± 5.83	49.58 ± 7.19	49.33 ± 5.91	49.13 ± 6.34
Electrical conductivity	S/m	1460.16 ± 240.29	1492.86 ± 186.64	1476.91 ± 219.96	1434.41 ± 194.74	1470.83 ± 166.82
TDS	Ppm	20,531.25 ± 317.55	21,670.59 ± 287.65	22,794.35 ± 353.94	23,436.16 ± 326.84	22,956.77 ± 367.81
Hardness	Ppm	116.82 ± 16.34	152.64 ± 14.27	146.21 ± 13.94	157.56 ± 15.37	136.99 ± 17.28
Alkalinity	Mg/L	84.25 ± 14.84	90.75 ± 11.58	95.29 ± 17.01	103.75 ± 14.32	98.51 ± 9.87
Total N	Mg/L	20.61 ± 2.64	21.73 ± 3.14	22.83 ± 2.95	24.15 ± 3.67	23.18 ± 2.97
Total P	Mg/L	0.66 ± 0.02	0.71 ± 0.11	0.74 ± 0.21	0.76 ± 0.29	0.74 ± 0.33
Plankton density	Ind/L	814 ± 32.87	1256 ± 26.52	965 ± 41.32	765 ± 22.59	893 ± 27.69

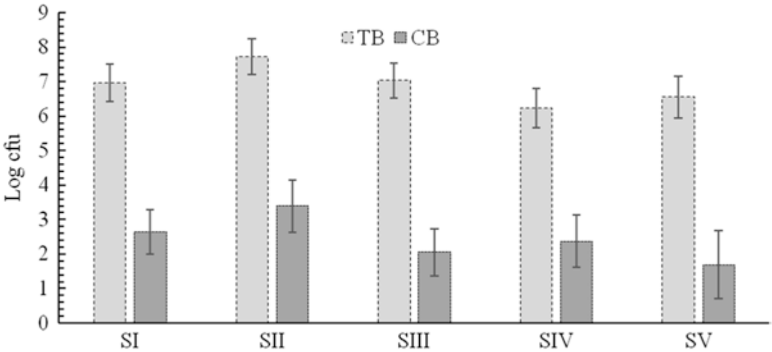

Fig. 5.1 Bacterial abundance associated with copepod, where total bacteria (TB) and chitinolytic bacteria (CB). Error bars represent the standard deviation where $n = 3$. *Indicates the significance of test using Tukey test at $p = 0.05$

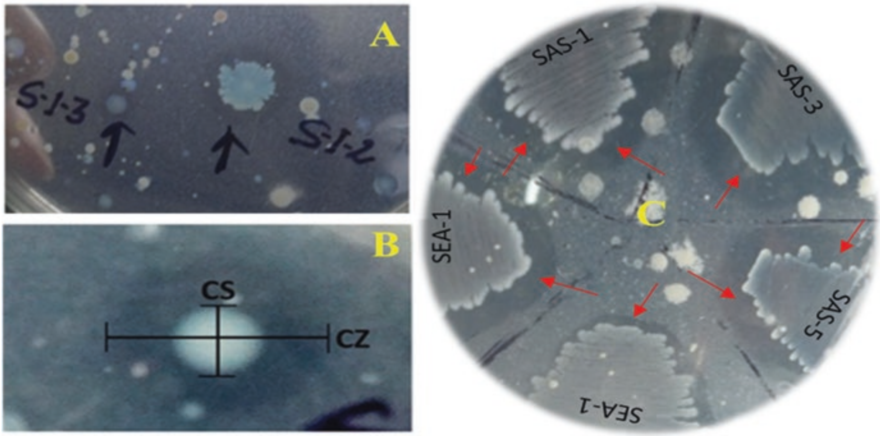

Fig. 5.2 Screening and selection of chitinolytic bacteria from plankton appendages (A) Primary isolation, (B) showing the colony size (CS) and chitinolytic zone (CZ) ratio, (C) secondary screening on collidalchitine agar (CCA) depending on their chitinolytic activity

chitinolytic index with colony size value >1 or equivalent to 1 revealed a potentially significant enzyme activity (Duncan et al. 2006). In the present study, all isolates showed chitinolytic activity with their CZ/CS ratio in between 1.41 and 1.75 pointing to the marked chitinolytic activity of these strains.

3.3 Biochemical Characterizations of the Isolates

Selected bacterial isolates that were grown in liquid media for biochemical identification of isolates reckon the absolute growth of microbes. To detect the biochemical responses of newly isolated bacteria, isolates were identified through an automated Vitek-2 compact (BioMérieux) system. From the Vitek 2 compact system data, it was concluded that nearly 62% isolates [SAS1 to SAS5] displayed the same biochemical activity out of total isolates, whereas the rest 48% isolates [SEA1 to SEA3] had more opportunity to grow at diverse conditions. It also revealed that isolates such as SAS1, SAS2, SAS3, SAS4, and SAS5 showed their abilities to grow in the media having six different sugar molecules, that is, glucose, maltose, mannitol, mannose, sorbitol, and trehalose, along with lipase activity (Table 5.2). These isolates were identified by series of biochemical tests such as sugar

Table 5.2 Sugar fermentation and biochemical assay of newly isolated bacteria SAS1–5 and SEA1–3 using Vitek-2 compact system isolated from the zooplankton appendages

S. No.	Biochemical assay	SAS1	SAS2	SAS3	SAS4	SAS5	SEA1	SEA2	SEA3
1	Adonitol	–	–	–	–	–	+	+	+
2	L-Arabitol	–	–	–	–	–	–	–	–
3	D-Cellobiose	–	–	–	–	–	+	+	+
4	H_2S production	–	–	–	–	–	–	–	–
5	D-glucose	+	+	+	+	+	+	+	+
6	D-maltose	+	+	+	+	+	+	+	+
7	D-mannitol	+	+	+	+	+	+	+	+
8	D-mannose	+	+	+	+	+	+	+	+
9	Lipase	+	+	+	+	+	–	–	–
10	Palatinose	–	–	–	–	–	+	+	+
11	Urease	–	–	–	–	–	–	–	–
12	D-sorbitol	+	+	+	+	+	+	+	+
13	D-tagatose	–	–	–	–	–	+	+	+
14	D-Trehalose	+	+	+	+	+	+	+	+
15	Citrate (sodium)	–	–	–	–	–	+	+	+
16	Malonate	–	–	–	–	–	+	+	+
17	L-lactate alkalinization	–	–	–	–	–	–	–	–
18	Succinate alakalinization	–	–	–	–	–	–	–	–
19	Phosphatase	–	–	–	–	–	+	+	+
20	Ornithine decarboxylase	–	–	–	–	–	+	+	+
21	Lysine decarboxylase	–	–	–	–	–	+	+	+
22	L-histidine assimilation	–	–	–	–	–	–	–	–
23	L-malate assimilation	–	–	–	–	–	–	–	–
24	L-lactate assimilation	–	–	–	–	–	–	–	–

+ and − denotes presence and absence of sensitivity towards these biochemical activities respectively.

Table 5.3 Sensitivity of newly bacterial isolates (SAS1–5 and SEA1–3) toward several isozymes of arylamidase, galactosidase, glucosaminidase, glucosidase, transferase, and xylosidase using Vitek 2 compact system isolated from the zooplankton appendages

Sl No.	Isoenzyme	SAS1	SAS2	SAS3	SAS4	SAS5	SEA1	SEA2	SEA3
1	Ala-Phe-pro-Arylamidase	–	–	–	–	–	–	–	–
2	L-Pyrollidionyl arylamidase	–	–	–	–	–	–	–	–
3	Beta galactosidase	–	–	–	–	–	+	+	+
4	Beta-N-acetyl glucosaminidase	+	+	+	+	+	+	+	+
5	Glutamyl arylamidase	–	–	–	–	–	+	+	+
6	Gamma-glutamyl transferase	–	–	–	–	–	+	+	+
7	Beta-glucosidase	–	–	–	–	–	+	+	+
8	Beta-xyloasidase	–	–	–	–	–	+	+	+
9	Beta-alanine arylamidase	–	–	–	–	–	–	–	–
10	L-proline arylamidase	+	+	+	+	+	–	–	–
11	Tyrosine arylamidase	–	–	–	–	–	–	–	–
12	Alpha-glucosidase	–	–	–	–	–	–	–	–
13	Alpha-galactosidase	–	–	–	–	–	+	+	+
14	Glycine arylamidase	–	–	–	–	–	–	–	–
15	Beta-glucuronidase	–	–	–	–	–	–	–	–
16	Lu-Gly-Arg-Arylamidase	–	–	–	–	–	–	–	–

+ and – denote presence and absence of sensitivity toward these enzymes respectively.

fermentation, production of gas and acetoin from glucose, lipase activity, and fermentation of sorbitol and trehalose (Bryant et al. 1986; Janda et al. 1984; Kuijper et al. 1989). Isolates SEA1, SEA2, and SEA3 were grown at 10 different sugar molecules, that is, adonitol, cellobiose, tagatose, palatinose, and other six as previously stated. These SEA isolates also showed ornithine decarboxylase, lysine decarboxylase, and phosphatase activity (Table 5.3).

3.4 *Isoenzyme Activity Analysis of the Isolates*

Vitek2 system characterizations of isolates obtained from the samples of crustacean zooplankton copepods have revealed sensitivity toward isozymes for Beta- N-acetyl Glucosaminidase activity of all isolates. The bacterial strains such as SEA1, SEA2, and SEA3 have been found to be sensitive toward other isozymes like alpha-galactosidase, beta-galactosidase, beta-glucosidase, beta-xyloasidase, glutamyl aryl amidase, and gamma-glutamyl transferase. The bacterial isolates, namely, sas1, sas2, sas3, sas4, and sas5 have revealed sensitivity to L-Proline Aryl amidase only. A negative response was found toward the isozymes of Alpha-Glucosidase, Ala-Phe-Pro-Aryl amidase, Beta-Alanine Arylamidase, Beta-Glucuronidase, Glycine

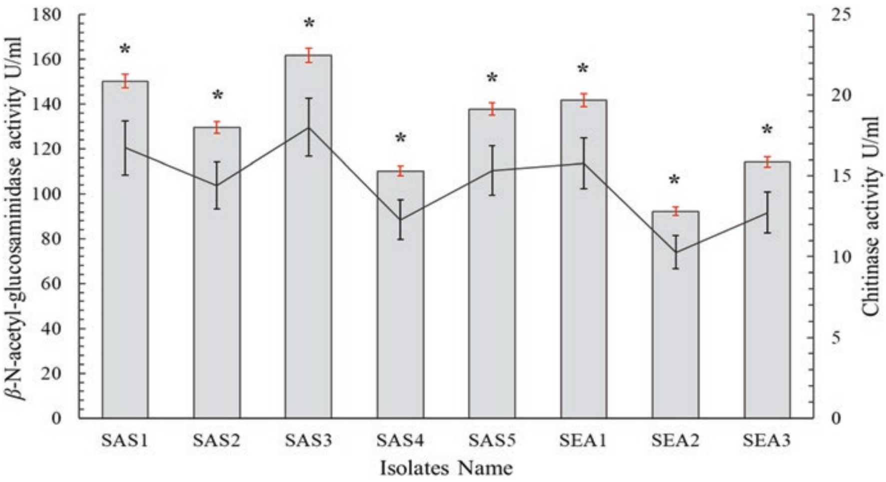

Fig. 5.3 Enzyme activity optimization of isolates SAS1 to SAS5 and SEA1 to SEA3. Error bars represent the SD values of triplicates. *Shows significance of the data at $p = 0.05$

Aryl amidase, L-Pyrollidionyl Arylamidase, Lu-Gly-Arg-Arylamidase, and Tyrosine Aryl amidase for all the native isolates (Table 5.3).

3.5 *β-N-Acetyl-Glucosaminidase and Chitinase Activity*

The enzyme chitinase is found to possess two major groups, that is, endochitinases (EC 3.2.1.14) and exochitinases, where exochitinases are further subdivided into two more groups—chitobiosidases (EC 3.2.1.29) and β-(1,4) N-acetyl glucosaminidases (EC 3.2.1.30) (Neetu et al. 2006). The enzyme *β*-(1,4) N-acetyl glucosaminidases can break down the oligomeric chain of chitobiosidases and endochitinase from substrates at the nonreducing end, producing low molecular monomers of GlcNAc (Neetu et al. 2006; McCreath and Graham 1992). In a favorable environment, the bacterial isolate, SAS3, could produce maximum enzyme activity, that is, 161.7 U/mL, whereas the bacterial isolate, SEA2 exhibited the lowest enzyme activity, that is, 92.3 U/mL toward the *β*-N-acetyl-glucosaminidase enzyme activity, with chitinase activity 18.02 U/mL and 10.27 U/mL, respectively (Fig. 5.3). From the abovementioned results, it was concluded that bacterial isolates SAS tend to display better enzyme catalytic effects than SEA isolates. Correlation study also revealed a significantly positive relationship between the enzymes *β*-N-acetyl-glucosaminidase and chitinase, that is ($P < 0.05$, $r = 0.359$), which indicates that the formation of monomers of GlcNAc continuously takes place during substrate enzyme catalytic reaction.

3.6 Identification of Isolates and Phylogenetic Tree Analysis

The evolutionary time was inferred using the neighbor-joining method (Saitou and Nei 1987). The ideal tree with the sum of branch distance 0.05201051 and 3.29384663 was shown (next to the branches), respectively. The evolutionary detachments were calculated using the MCL method (Tamura et al. 2004) and are in the units of the number of base replacements per site. The investigation involved 11 and 9 nucleotide sequences respectively, and codon locations included were 1st + 2nd +3rd + Noncoding. All sites containing gaps and missing data were eliminated; there were a total of 1414 and 1286 positions, respectively, in the final dataset. Evolutionary analyses were shown in MEGA7 (Kumar et al. 2016). The phylogenetic analysis revealed that the bacterial isolates, such as SAS1, SAS2, SAS3, SAS4, and SAS5, belonged to *A. sobria* (Fig. 5.4), and the bacterial isolates such as SEA1, SEA2, and SEA3 were associated with Enterobacter aerogenes (Fig. 5.4), (Jung and Jung-Schroers 2011; Donderski and Trzebiatowska 1999). A previous study about chitinolytic bacteria advocated that the occurrences of various bacterial species, that is, Enterobacter aerogenes, *Aeromonas spp,* and *Vibrio spp,* have shown their chitin degradation activity (Tang et al. 2001; Swiontek and Donderski 2006; Huang et al. 1996; Russell et al. 1988).

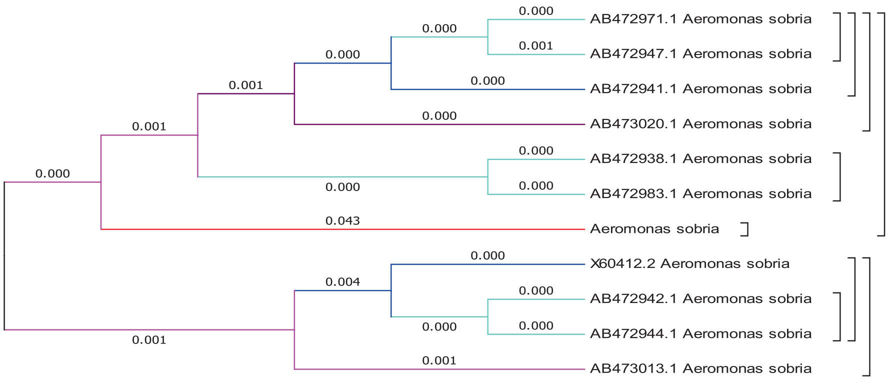

Fig. 5.4 (**a**) Phylogenetic tree analysis of 11 taxa generated by comparing 16S rDNA homology in MEGA-7 software, showing the location of newly isolates strain. The GenBank accession numbers are shown in parentheses. (**b**) Phylogenetic tree analysis of 9 taxa generated by comparing 16S rDNA homology in MEGA-7 software, showing the location of newly isolates strain. The GenBank accession numbers are shown in parentheses

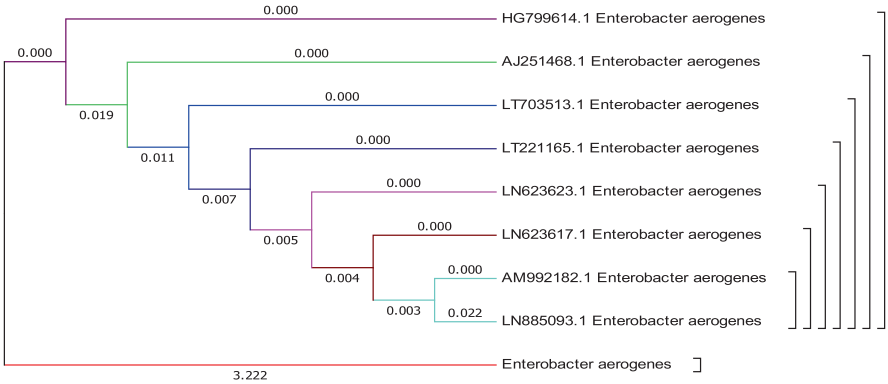

Fig. 5.4 (continued)

4 Discussion

The community of Zooplankton is considered to be a vital component of the aquatic ecosystem, and it controls and maintains the aquatic food webs. Thus, its positioning in the food chain of the system with its high degree of association to the primary producers makes it extremely susceptible to different structural heterogeneity in the system. Chitinolytic bacteria have played an important function in the aquatic ecosystem based on their extensive involvement in energy flow and nutrient cycling (Sviti et al. 1997). A number of studies have confirmed the importance of microorganisms, particularly bacteria, as major sources of chitinase, which have been in use for several biomedical applications for antibacterial and antifungal treatments and also in the foods processing industries (Bhattacharya et al. 2007; Singh et al. 1999). The exoskeletons of crustaceans, including the copepod zooplankton, are mainly constituted by chitin and act as major sources of nutrients (N, C, etc) in the aquatic environment on their decomposition by mainly chitinolytic of various types of bacteria. This autochthonous (within the aquatic ecosystems) mode of nutrients generation along with allochthonous ones (supplied from outside) trigger the trophic interactions as well as energy flows within the aquatic ecosystem (Nagasawa and Nemoto 1988; Bhushan and Hoondal 1998; Kirchman and White 1999; Dahiya et al. 2005). Besides, the total bacterial population and total chitinolytic bacterial population as recorded in the present study incited the health risk possibility by the propagation of bacteria getting support from the zooplankton in respect of shelter and nutrients. A study about the relationship between bacteria and copepod interaction also agreed that highly bacterial abundances were carried out by various copepods (Bomar 1988; Shoemaker and Moisander 2015, 2017; Eswaran and Khandeparker 2019; Dalvin et al. 2020). Out of a total of eight bacterial species isolated from zooplanktonic copepod *Heliodiaptomusviduus,* two different bacterial genera, namely, *Aeromonas spp.* and *Enterobacter spp.*, were observed. Earlier researchers have reported that these two genera are consortia together with zooplankton in the aquatic environment (Veronica and Kalisa 2019; Midya et al. 2019). Both isolated genera displayed comfortable growth utilizing different carbohydrate sources like glucose, maltose, mannitol, mannose, sorbitol, and trehalose, where *Enterobacter* genera showed some different characteristics features, which also grows in some other sugar molecule, that is, tagatose, palatinose, adonitol, and cellobiose. Our findings, growing condition in presence of different carbon sources, are compiled with the previously published report (Sharon et al. 2003; Gowhar et al. 2016; Ravea et al. 2018). All the isolated genera have the activity toward the chitinase and Beta- N-acetyl Glucosaminidase enzymes. These two enzymes play a decisive role in the nitrogen and carbon cycle in the aquatic ecosystems, where these enzymes are N-targeting hydrolytic enzymes also release the carbon from the substrate for Beta- N-acetyl Glucosaminidase is chitin (Ling et al. 2017; Paulsen et al. 2016; Arnostic 2011). In the present investigation, a correlation study has revealed a significantly positive relationship between the enzyme of β-N-acetyl-glucosaminidase and chitinase ($P < 0.05$, $r = 0.359$), which indicates that the formation of monomers of GlcNAc continuously takes place during substrate enzyme catalytic reaction. The previous researcher also advocates that there was a positive correlation in Beta-N-acetyl Glucosaminidase activity and nitrogen mineralization and found that Beta-N-acetyl Glucosaminidase enzyme activity decreases in high nitrogen availability and that enzyme plays a crucial role in nitrogen cycling (Ekenler and Tabatabai 2002, 2004; Min et al. 2011). However, the effect of chitinase-producing bacteria on plankton containing chitin is unclear, which has necessitated undertaking the present study to highlight the interaction between chitinase-producing bacteria and zooplankton, and the result outcomes are expected to open up new vistas in the ecological research of aquatic ecosystem on one hand and epidemiological awareness developments toward the right usage of surface water bodies, especially in a developing country having lack of proper sanitation facilities, value in the protection, and utilization of freshwater plankton resources. The 2- D plot of PCA analysis was generated according to the first two components (component 1 and component 2) which reveals that the component 1 had an eigenvalue of 6.829 and 48.778% variable coverage, whereas component 2 had an eigenvalue of 4.649 and 33.209% variance coverage (Fig. 5.5). According to the rotated component matrix, water parameters like pH (−0.872), dissolved oxygen (−0.977), Secchi depth (−0.824), and temperature (−0.888) were closely related and negatively correlated with the total bacteria (0.910), chitinolytic bacteria (0.828), and plankton density (0.890). The negative influence of DO, pH, and temperature on bacterial abundance parameters during study times agrees with a maximum bacterial density that causes the exhaustion of DO that can be anticipated to the maximum rate of bacterial decomposition of the extreme organic substance discharged unswervingly into the water body (Singh et al. 2002). On the other hand, other water parameter factors like biological oxygen demand (0.997), electrical conductivity (0.691), total dissolved solids (0.908), total nitrogen (0.824), total phosphate (0.928), and hardness (0.849) were closely related and positively correlated with the plankton density, total bacteria, and chitinolytic bacteria. High TDS, TN, TP, and BOD values were also supported the copious propagation of bacteria in these study

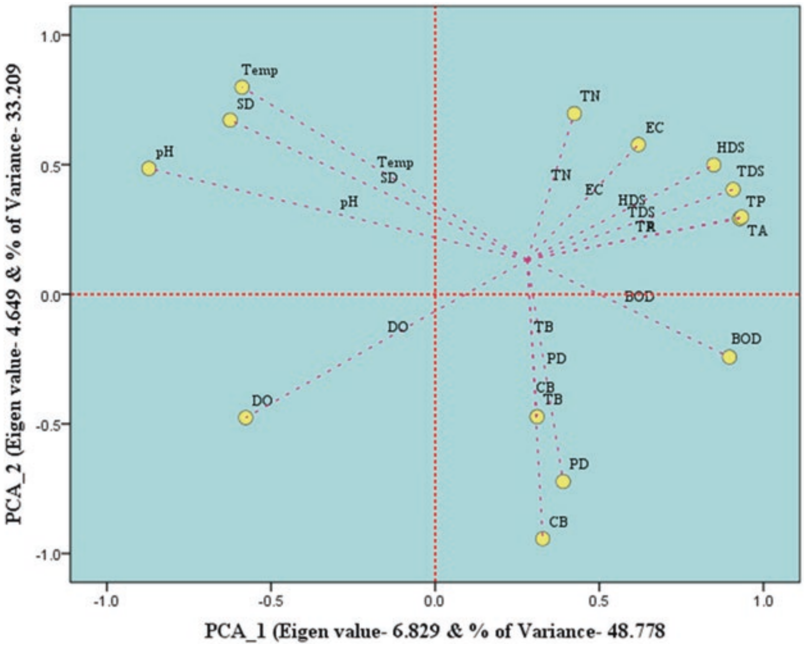

Fig. 5.5 Principal component analysis (PCA) showing the relationship between plankton density, total bacteria, and total chitinolytic bacteria with water physiochemical parameter. Where, *Temp*. temperature, *SD* Secchi depth, *DO* dissolve oxygen, *TN* total nitrogen, *TP* total phosphate, *EC* electrical conductivity, *HDS* hardness, *TDS* total dissolve solids, *TA* total alkalinity, *BOD* biological oxygen demand, *TB* total bacteria, *CB* chitinolytic bacteria and *PD* plankton density

sites since BOD is a pollution indicator. So, the relationship between bacterial density and BOD suggests that at high organic loading rates, the aquatic ecosystem favors the multiplication of anaerobic microbial population (Mtui and Nakamurs 2006).

5 Conclusion

The observation of the zooplankton and microbe's species assemblage of riverine wetlands indicates considerable diversity in that region. Abundance of Zooplankton showed varied seasonal differences between the natural site and industrial adjoin region of the river. Anthropogenic influence and industrial effluent may cause higher density of bacteria at site-II and dominance of pollution-tolerant macrophytes with zooplankton species in site-II. Also, different known indicator species of zooplankton showed dominance in site-II. These findings would be helpful in planning a more suitable management strategy of Paschim Medinipur wetlands to ensure the sustainability of the fishery industry and also conservation of natural resources. The outcome of the present research has revealed that both the chitinolytic bacterial strains *A. sobria* and Enterobacter aerogenes were novel sources of chitinase enzymes which not only played a crucial role in the decomposition of chitin to release a good amount of nutrients within the aquatic ecosystem but also to utilize the abundances of the copepod zooplankton, *Heliodiaptomusviduus*, for their own proliferation.

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Chapter 6
Microplastics in Freshwater Riverine Systems: Brief Profile, Trophic-Level Transfer and Probable Remediation

Urmi Mustafi Moon, Chhandak Mondal, Nimai Chandra Saha, and Asif Hossain

Abstract The story of human civilization will remain incomplete without addressing the contribution of rivers in our life. The riverine system is the largest source of the freshwater aquatic body. River confluences and its tributaries play a significant role in human life. Today, the riverine systems from all over the world especially the South-Asian rivers are moving towards an alarming situation due to anthropogenic pollution. South Asia, the cultural and economically developing part of the Asian continent, is now combating successive increases in high population rate, urbanization and industrialization. All these factors are cumulatively leading towards riverine pollution. Plastics, especially microplastics pollution, are one of the major threats to the degradation of riverine health due to high dependence on plastics to meet expectations of daily life. Studies from different sources reveal that most of the rivers of the world are getting polluted by microplastics (<5 mm in diameter). Natural weathering processes transform the plastics into microplastics which pose a serious threat to the aquatic biota and food chains. Remediation strategies are still less understood for this new-age pollutant. This article is an initiative to gather data on plastics and microplastics pollution in the riverine system and a scientific overview of the present situation.

Keywords Freshwater · River · Microplastics · Pollution · Food chain

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1 Introduction

The journey of human civilization started on the bank of river. Historical evidences suggest that from the great Egyptian Civilization to the Indus Valley Civilization and almost all the great civilizations of the earth put their first cornerstone on the bank of the river. The contributions of rivers and riverine systems in our life are immense but in exchange, human civilization has contributed largely by enhancing pollution. Pollution in riverine systems has now become a global phenomenon. Rapid economic development, urbanization, industrialization and high population growth rate are considered as the major causes of river pollution throughout the world (Shruti et al. 2019; Kieu-Le et al. 2019). Other factors are oil and natural gas exploration, emission of chemical and industrial effluents (Mani et al. 2016), garbage dumping (Fok and Cheung 2015) and use of pesticides (Chopra et al. 2011) in agricultural fields; all these activities are the cumulative causes of riverine pollution. Many rivers across the world are highly polluted with domestic, industrial and agricultural wastes. As a result, the Water Quality Index (WQI) is reaching unsafe levels (Shukla et al. 2018; Zhang et al. 2010). The analysis of daily variability in litter type shows that most of the materials are household polymer material, rubber, chemicals, cloth, glass, ceramics, plastic cups and so on.

Recently, scientists are focusing on plastic pollution, mainly microplastic and nanoplastic pollution with prime concerns. The global annual production of plastics was 1.7 mt (million tons) in 1950 rising massively to 311 mt in 2014 (Polska 2015). Plastic is considered as one of the most man-made nuisance material. It achieves its suitable position in our society due to its cheap, flimsy, durable and flexible characteristics. These nonbiodegradable or less biodegradable materials persevere in the soil interminably. Gradual increase in the amount of plastic pollution inside the freshwater systems as well as in the marine systems have become a severe headache and many rivers across the earth are now combating with declining water quality due plastic pollution (Loucks et al. 2005; Wagner et al. 2014; Zhang et al. 2015). Scientific investigation reveals that about 1.15–2.4 mt of plastics debri are being deposited into riverine systems every year (Lebreton et al. 2017) and 80% of plastic debris which is emitted by rivers into the oceans actually originated from inland sources (Jambeck et al. 2015). Recent studies on global riverine systems indicate that Asia, mostly the South-Asian rivers, acts as an epicenter of riverine plastic pollution and Australia–pacific (0.02%), Europe (0.28%), Central and North America (0.95%), South America (4.8%), and Africa (7.8%) contribute accordingly (Australia, National Water Commission 2007). Empirical studies show that each year about 8 mt of plastics is deposited into the Pacific Ocean through watersheds from different parts of the world (Virsek et al. 2017; Rios-Mendoza et al. 2018). In a prediction, the UNEP (Kershaw 2015) expecting that weight of plastics in the ocean will be more than the weight of fish by 2050.

Weathering of plastics due to physical, chemical or biological routes results in small fragments, ultimately resulting in the formation of microplastics (<5 nm diameter) (Law and Thompson 2014). Several studies on riverine pollution disclose that microplastics (MPs) hits the different parts of the global riverine systems (Lahens et al. 2018) and this lesser sized plastics can be easily ingested by zooplanktons, birds, fishes, shellfishes and other micro- and macro-benthos and different vertebrate organisms creating several adverse effects on the food chains (Van Cauwenberghe and Janssen 2014; Auta et al. 2017; Duncan et al. 2019).

The present study is a review work to understand the contemporary scenario of plastics, mostly the MPs pollution, their sources and consequences on aquatic biota in the riverine systems and the possible management strategies.

2 Major Pollutants Responsible for Degradation of Riverine Health

Rivers give out many social activities and are also intensively subjective to ecosystems. Ecosystem services of the rivers for sustainable developments to the human civilization have weakened in recent times. The assessment of river health as measured by the function of physical, chemical and biological activities needed to preserve the organization and ecosystem functioning of the river.

Riverine health is being disturbed by the different types of anthropogenic activities and pollutants. The accidental spillage of oil and natural gases (polycyclic aromatic hydrocarbon and lipophilic organic contaminant etc.) from the large vessels, ships and tankers into the river water (Zhang et al. 2012; Schwarzbauer and Heim 2005) expose disastrous transformation in the aquatic food web, public health and other ecosystem functions. Discharge of the industrial and chemical effluents (Phiri et al. 2005; Rajaram and Das 2008), heavy metals (Cooper 1993) and agricultural runoff (Hussain et al. 2017; Ali et al. 2016) are the three main sources upsetting the water quality.

3 Microplastics as an Anthropogenic Pollutant

Leo Baekeland invented Bakelite, the first fully synthetic plastic in the year 1907. Over the last 50 years, plastics have been saturating the earth's ecosphere. The United States increased plastic production up to 300% during World War II (Thompson et al. 2009). Plastics become more hazardous and more harmful when it is converted into MPs. Under the process of physical, mechanical weathering, exposure of UV light and the actions of waves, plastic litter disintegrates and breaks down into smaller particles, such as fibers, granules, flakes, spheroids, fragments, beads and pellets (von Moos et al. 2012; CONTAM 2016). Plastic particles that fall in the range between 0.1 μm to 5000 μm are called MPs (Lusher et al. 2017; Arthur et al. 2008).

Table 6.1 Categorization of micro plastic particles (MPs). (Lusher et al. 2017)

Types	Examples
Fragments	Irregular shaped particles, tuffs, crystals, granules, flakes, etc.
Fibers	Filaments, threads, microfibers, strands, etc.
Beads	Grains, strands, spherical microbeads, threads, microspheres, etc.
Pellets	Pre-production pellets, resin pellets, nibs, nurdles, etc.

Microplastics are divided into two major groups: Primary Microplastics and Secondary microplastics. **Primary MPs** are plastic particles with <5 mm diameter in size (Kershaw and Rochman 2015). It includes (a) preproduction resin pellets (3–5 mm), (b) microbeads, and (c) beads. Primary MPs are found in the novel close-to-novel structure during their collection (Driedger et al. 2015). Secondary MPs are the derivatives of the disintegrated and weathered large volume of plastics. The generation of secondary MPs takes place when primary MPs break down to form smaller pieces. Secondary MPs have been reported to originate from terrestrial and marine-based resources. Marine-based resources include sewage from ships, fishing equipment and so on, whereas terrestrial-based resources include polythene baggage, packing material and so on (CONTAM 2016). Depending on different shapes, MPs can be categorized into fragments, foams, fibers, beads and pellets which can be tabulated in the following manner (Table 6.1).

MPs originate from a range of organic polymers such as polystyrene (PS), polyethylene (PE), polyurethane (PU), polyvinylchloride (PVC), PE terephthalate (PET) and polypropylene (PP) and so on. MPs act as potential vectors in transporting hydrophobic organic chemicals (HOC) from the aquatic environment (Koelmans et al. 2016) that pose a serious threat on aquatic organisms including fish (Dantas et al. 2012), crab (Lebreton et al. 2017; Watts et al. 2014; Farrell and Nelson 2013), mussels (Paul-Pont et al. 2016; Avio et al. 2015) and so on. Observing their persistence ability and harmful effect on the environment, these are commonly termed as persistent organic pollutant (POP). Moreover, MPs acts as a carrier of antibiotics in the aquatic environment imposing negative effect to the aquatic food chain (Li et al. 2018).

4 Distribution of MPs in Aquatic Ecosystems

Every year a huge number of plastic particles are making their way towards different aquatic systems and the marine ecosystem is the ultimate reservoir of plastic litters. Estimation over plastic pollution brings out the fact that about 4.8–12.7 MT (metric tons) of mismanaged plastic debris entered into the marine system in 2010 (Jambeck et al. 2015). About 10% of all plastic waste ultimately mixes up with ocean water every year (Thompson 2015). The presence of MPs was first reported in planktons throughout the coastal areas of New England in North America in 1970 (Carpenter and Smith 1972). We will be surprised to notice that MPs can be found even in the most remote places of the earth as in the deep sea (van Cauwenberghe et al. 2013), the Tibet Plateau (Zhang et al. 2016) and the Arctic zone (Obbard et al. 2014) evidencing the omnipotence distribution of MP particles. Recent studies reveal that most of the freshwater aquatic body, estuarine system and marine environment are now suffering from plastics and MPs pollution because it can accumulate on shoreline, sea surface, sediment and natural habitats (Thompson et al. 2009; Ryan et al. 2009; Moore 2008; Barnes et al. 2009).

Empirical investigations on aquatic ecosystems reflect that most of the marine and freshwater aquatic system such as rivers (Moore et al. 2011; Horton et al. 2017; Morritt et al. 2014; Miller et al. 2017) and lakes (Eriksen et al. 2013; Imhof et al. 2013; Faure et al. 2012; Free et al. 2014; Eriksen et al. 2013) of the world are facing this unhealthy situation (Table 6.2). Superior, the largest freshwater lake on the earth grabs the attention of scientists for the presence of MPs in its aquatic system. Examining the surface water of Lake Superior, North America, 463,423 items have been detected per kilometer (Eriksen et al. 2013) as maximum. Studies on the sediment and surface water of Lake Garda, Italy (Imhof et al. 2013) Lake Geneva, Europe (Faure et al. 2012) and Lake Hövsgöl, Mongolia (Free et al. 2014) show the presence of MP particles in every sample. The Tamar estuary, United Kingdom (Sadri and Thompson 2014), Chesapeake Bay, United States (Yonkos et al. 2014) and Pearl River estuary, China (Fok and Cheung 2015) show the evidence of MP particles in their water sample. The major riverine systems of Asia and Africa (Peng et al. 2018; Alam et al. 2019) are struggling with this unavoidable situation where the North and South America (Shruti et al. 2019; Moore et al. 2011) and European riverine basin (Horton et al. 2017; Morritt et al. 2014) are not exceptions. The distribution of plastic particles into the ocean, sea bed and shoreline depends on the coastline geography, water current, surface circulation, wind velocity and proximity of the coastal area to the urban region (Kang et al. 2015; Corcoran 2015). Floating plastic debris which also includes MPs was reported in the subtropical gyres in early 1970 in the North Atlantic (Colton et al. 1974; Law et al. 2010), North Pacific (Moore et al. 2001), South Pacific (Eriksen et al. 2013) and outside the subtropical gyres of near-shore environment (Thompson 2004) (Table 6.3.).

5 Freshwater Riverine Systems are in Crisis due to MPs Pollution

Rivers are the ultimate integrations of watershed hydrology. In the last few decades, the occurrence of MP in the oceanic environment has been described throughout the world (Browne et al. 2011; Ivar do Sul and Costa 2014; Waller et al. 2017) and the consequences of MP pollution in freshwater food web has been emphasized in the recent times (Duis and Coors 2016; Su et al. 2018; Horton et al. 2017). Earlier

Table 6.2 Some ecologically important fresh water aquatic systems with huge MPs contamination (Eerkes-Medarno et al. 2015)

Aquatic systems	Test	Size classes	Abundance	References
Los Angeles river	Water	> ¼ 1.0 and < 4.75 mm, >¼4.75 mm	Max:12,932 items m^{-3} Mean 24-h particle counts	Moore et al. (2011)
Lake Superior	Water	0.355–0.999 mm, 1.00–4.749 mm, >4.75 mm	Max: 463,423 items km^{-2} Mean: 43,157 items km^{-2}	Eriksen et al. (2013)
Lake Garda	Sediment	9–500 μm, 500 μm to 1 mm, 1–5 mm and > 5 mm	Max: 1108 ± 983 items m^{-2} Mean: not indicated	Imhof et al. (2013)
Lake Geneva	Water and sediment	Sediments: 81% represents size <4.75 mm < 2 mm, < 5 mm Water: <5 mm, >5 mm	Sediment: Max: 9 items m^{-3} Water: Max: 48–146 items km^{-2} Mean: not indicated	Faure et al. (2015)
Lake Hövsgöl	Water	0.355–0.999 mm, 1.00–4.749 mm	Max: 44,435 items km^{-2} Mean: 20,264 items km^{-2}	Free et al. (2014)
Laurentian Great Lakes	Water	0.355–0.999 mm 1.000–4.749 mm > 4.75 mm	Max: 43,157 ± 115,519 km^{-2} Mean: not indicated	Eriksen et al. (2013)
Lake Huron	Sediment	<5 mm, >5 mm	Total pieces: 3209 Mean: not indicated	Zbyszewski and Corcoran (2011)
Lake Erie	Sediment	<2 cm	Total pieces:1576 Mean: not indicated	Zbyszewski and Corcoran (2011)
Danube river	Water	<2 mm, 2–20 mm	Max: 141–647.7 items 1000 m^{-3}, Mean: 316.8 (±4664.6) items 1000 m^3	Lechner et al. (2014)
St. Lawrence river	Sediment	Not indicated	Max: not indicated Mean: 13759 (±13,685) items m^{-2}	Castaneda et al. (2014)

studies suggest MPs appeared to prevalently exist in the sediments and surface water of many freshwater environments in Europe (Fischer et al. 2016; Klein et al. 2015), North America (Driedger et al. 2015; Sadri and Thompson 2014; Baldwin et al. 2016), Africa (Heskett et al. 2012), Australia (Gregory 1977), Asia (Eo et al. 2018), China (Su et al. 2016; Wang et al. 2017; Zhang et al. 2010) and South Korea (Chae et al. 2015; Song et al. 2015). Study on the global scenario of mismanaged waste generation and emission of plastic marine debris into the aquatic body, upon twenty different countries across Asia, Africa, South America and the United States continents discloses that six Asian member countries generate 80% of the

Table 6.3 Comprehensive list of locations in marine system detecting MP particles (Nerland et al. 2014)

Location	Avg. concentration (particles)	Citations
Coastal areas of North-west Atlantic Ocean	$3/m^{-3}$	Carpenter and Smith (1972)
Offshore areas of North-west Atlantic Ocean	$67/m^{-2}$	Law et al. (2010)
Central gyre of North Pacific Ocean	$334.3/m^{-2}$	Moore et al. (2001)
Celtic Sea of North-east Atlantic Ocean	$2.46 \pm 2.43/^{-3}$	Frias et al. (2014)
Portuguese coast areas of North-east Atlantic Ocean	$0.002–0.036/m^{-3}$	Collignon et al. (2012)
Sardinia of Western Mediterranean Sea	$0.116/m^{-2}$	Collignon et al. (2014)
Corsican areas of Western Mediterranean Sea	$0.062/m^{-2}$	Faure et al. (2015)
Western Mediterranean Sea	$130/m^{-2}$	de Lucia et al. (2014)
Central areas of Western Mediterranean Sea	$0.15/m^{-3}$	Moore et al. (2002)
Southern Californian areas of North-east Pacific Ocean	$8/m^{-3}$	Doyle et al. (2011)
North-east Pacific Ocean	$0.004–0.19/m^{-3}$	Goldstein and Goodwin (2013)
Subtropical gyre of North Pacific Ocean	$0.021–0.448\ /m^{-2}$	Nerland et al. (2014)
Subtropical gyre of North Pacific Ocean	$>1000\ /m^{-2}$	Wright et al. (2013)
Coastal areas of South Korea	13 ± 11 Ocean/m^{-2}	Eo et al. (2018)

mismanaged plastic waste into the river (Jambeck et al. 2015) (Table 6.4). South Africa (2%), Nigeria (2.7%), Egypt (3%), Thailand (3.2%), Vietnam (6%), Philippines (6%) and Indonesia (10%) are conscientious for more than half of the plastics entering the ocean followed by China (28%), which secure the highest position in releasing mismanaged plastic waste into the ocean (Kim et al. 2015) (Fig. 6.1).

6 Difficulties of South-Asian Rivers: An Overall Scenario of MPs Pollution

The cultural and economic backbone of South Asian countries largely depends on its great rivers, such as the Ganges, Indus, Yangtze, Mekong, Irrawaddy and so on. These great rivers are known historically for their contribution to the development and affluence of several earliest civilizations and supporting the livelihoods of millions in these highly populous areas of the world.

Rapid development and high population density in these areas leading to the environmental deterioration and riverine systems in this area are no exception. Like any other part of the earth, South Asian Rivers are suffering tremendously from the

Table 6.4 List of top 20 countries on mismanaged plastic waste estimate for 2010 (in units of millions of metric tons per year)

Rank	Country	Percentage of plastic waste	Percentage of mismanaged waste	Plastic Debris (MMT/Years)
1	China	11	76	1.32–3.53
2	Indonesia	11	83	0.48–1.29
3	Philippines	15	83	0.28–0.75
4	Vietnam	13	88	0.28–0.73
5	Sri Lanka	7	84	0.24–0.64
6	Thailand	12	75	0.15–0.41
7	Egypt	13	69	0.15–0.39
8	Malaysia	13	57	0.14–0.37
9	Nigeria	13	83	0.13–0.34
10	Bangladesh	8	89	0.12–0.31
11	South Africa	12	56	0.09–0.25
12	India	3	87	0.09–0.24
13	Algeria	12	60	0.08–0.21
14	Turkey	12	18	0.07–0.19
15	Pakistan	13	88	0.07–0.19
16	Brazil	16	11	0.07–0.19
17	Burma	17	89	0.07–0.18
18	Morocco	5	68	0.05–0.12
19	North Korea	9	90	0.05–0.12
20	United States	13	2	0.04–0.11

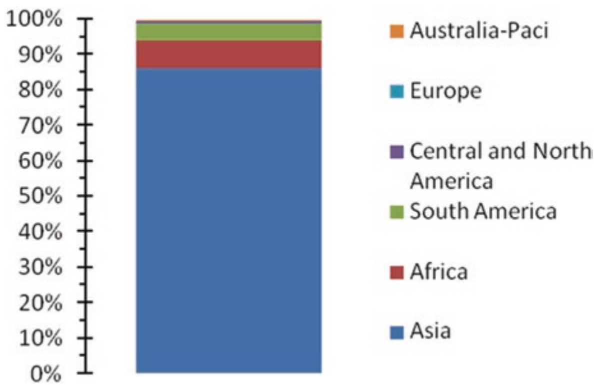

Fig. 6.1 Global river input of plastics to the ocean by different continents. (Lebreton et al. 2017)

mismanaged plastic waste generation. South Asian Rivers are the highest contributors of anthropogenic plastic debris into the ocean (Lebreton et al. 2017). The reasons behind this are the rapid economic and industrial development and the ever-increasing population growth of Asia. Empirical studies (Browne et al. 2011) established that there is a significant positive association between MP abundance and density of human population in the urban and suburban watersheds. Most of the

South-Asian megacities such as Hong-Kong, Shanghai and Ho-Chi-Minh city reside on the river bank and cities' wastewater including anthropogenic debris and mismanaged waste are discharged directly into the oceans. This altogether disrupts the ecological balance of aquatic systems. In this section, we are focusing on the unprecedented level of plastic waste in these regions of riverine systems.

Yangtze is the largest tributary in Asia and is the third largest in the globe. This river rises from the Tibetan plateau and has played a major part in the development of the culture, history, and wealth of China. Studies on the seven different sampling sites of Yangtze estuarine system reveals the presence of MP in all the sampling sites. The typical concentration of Micro Plastics in the Yangtze riverine system is reported as 4137.3 ± 2461.5 nm^{-3} (Ritchie and Roser 2020). *Pearl River* is China's third longest river and flows 2400 km across China and Vietnam, and the river water is extremely exploited by Hong-Kong City, with a daily waste of plastics of 1866 tons in the year 2013 (Zhao et al. 2014). This mismanagement of plastic debris leads towards an awful situation that the Pearl river estuary acts as the potential source of MP. Moreover, the combined input of Zhugiang, Dong, and Xie Rivers degrades the water quality making it unsuitable for aquatic animals. The anthropogenic debris of urban and suburban regions are the major causes of MP pollution in the riverine system. Shanghai, which is the most populated area of China covering 6340 km^2 and having 24.15 million populations (HKEPD 2015), contributes a large number of plastic into freshwater riverine systems of Huangpu, Shajinggang, Caohejing, Beishagang, Jiangjiagang, and Yujibang Rivers (Shanghai Statistical Yearbook 2016). The abundance of MPs in sediment samples of these six rivers were reported between 80.2 ± 59.4 items/100 kg dry weight (80.2 ± 59.4 kg/dw). All these evidences show the presence of MP in China's riverine system which is the biggest threat to China's aquatic ecosystem. After the river Yangtze, River Ganges is one of the largest rivers of South Asia and plays an essential function in upholding the cultural heritage of India and Bangladesh. The Ganges rises from the Gangotri glacier of Himalaya and empties into the Bay of Bengal securing the second top position as a polluting river (Lebreton et al. 2017) and discharges 2.08×10^4 (m^3/s) plastic particles annually. The reasons behind this are industrialization, high population density, people's religious belief, mismanagement of river water and so on. Other rivers that regulate the economic and cultural integrity of South Asia are River Brantas, Solo, Serayu of Indonesia, Pasig of Philippines, Irrawaddy of Myanmar and Mekong which is flowing over six countries. Over 5990 big and small rivers contribute to the surface water of Indonesia and Brantas is the largest river, located in the East Java province. Pollution in the surface water in Indonesia are chiefly because of poor hygiene, discharge of unprocessed sewage, short of sewage management and intermixing of municipal and industrial wastewater into the river (Peng et al. 2018). The annual discharge of plastics in river Brantas is 2.07×10^2 m^3/s (Lebreton et al. 2017) indicating a serious ecological degradation. The conditions of river Solo and Serayu of Indonesia are similar to the river Brantas. River Irrawaddy is the largest river of Myanmar and flows from north to south through Burma and empties into the Andaman Sea. This river is the most important commercial waterway of Myanmar. It supports a wide variety of both plant and animal species in its riverine ecosystem. The people of Myanmar highly depend on this river for irrigation, fishing and water supply and for domestic and cultural activities. This river is facing trouble by plastic pollution carrying 5.49×10^3 tons of plastic waste per year which leaves a notable mark on Myanmar's freshwater hydrology. River Mekong has an extensive network and webs over six Asian countries. It is Asia's seventh largest river and its path of 2703 miles is a cultural, industrial and economic growth zone of South Asia. This river supports a wide range of plant and animal species promoting a sustainable aquatic ecosystem. Plastic pollution in this major riverine system creates ecological imbalance and points out the failure of the administration in managing the plastic debris and stopping the degradation of water quality (Eo et al. 2018) in Mekong riverine system. River Saigaon is situated in southern portion of Vietnam that rises near Phum Duang in southeastern Cambodia, which is running through the Ho Chi Minh City, a developing megacity in South-East Asia. About eight million people inhabit this megacity and have a crucial role in the plastic waste discharge in the Saigaon riverine system (Le et al. 2016; Stardy et al. 2017). About 2000–13,000 MT (metric tons) of suspended waste is accumulated yearly from the riverine systems, which vividly indicates the city's poor management of plastic debris (Kieu-Le et al. 2019).

However, these altogether reflect the distressing scenario of plastic pollution in the South-Asian riverine system. Apart from South Asia, several instances have been found in the rest of the world. Estimated plastic release from African rivers is 7.80%, South American rivers is 4.80%, Central American and North American rivers is 0.95% and 0.28% from European rivers (Fig. 6.2).

7 MPs Pollution in the European River Basins

The abundance, distribution and density of MPs in the sediments of the freshwater riverine system across Europe have been studied by several investigators. Rhine river of Western Europe, culturally and historically one of the important rivers of the continent stretching across Germany, France, Switzerland, Netherland and Australia and ultimately empties into the North Sea, is flowing through a highly industrialized, dense region of western Germany and above 1000 companies that are related with the plastic industry. Researchers of the University of Basel, Switzerland, sampled 11 locations along 500 miles of the river's surface from Basel to Rotterdam and found that the water body has been chocked with MPs. They collected 25,956 MP particles from a water surface of 25,745 m^2. In another finding, the concentration of MPs in the Rhine River ranged from 228 to 3763 and 786 to 1368 items/kg, respectively (Klein et al. 2015). Thames, another important river of Europe, supports urban development, industrialization, tourism and receives sewage discharged by 13 million people throughout its distributaries (Stardy et al. 2017) and it is also suffering from huge plastic pollution in surface water as well as sediment system. It was reported that Po river of Italy and Seine river of France show the presence of MP particles in their water. MP samples collected using the manta trawl in Po River

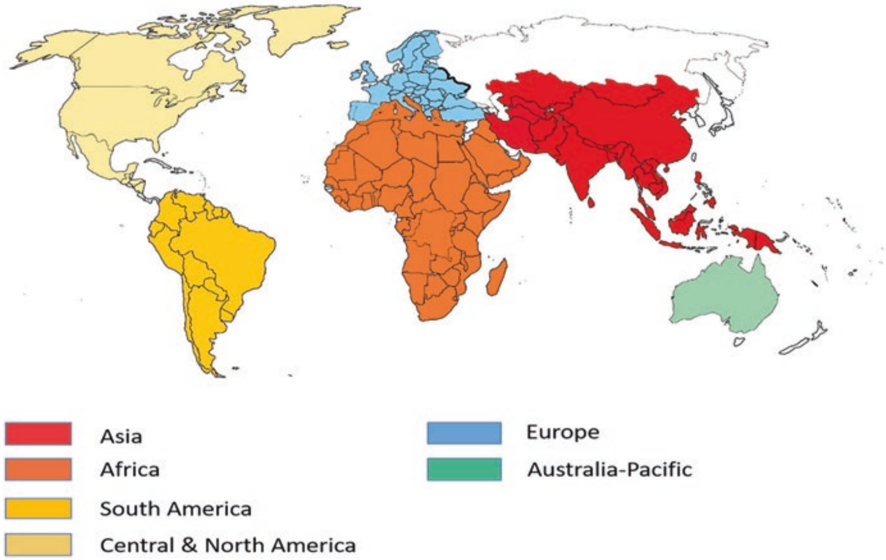

Fig. 6.2 Global share of mismanaged plastic waste generation. The colour gradient is used here to make understand the high, moderate and minimal plastic waste generation by the continents. The red colour signifies the highest contributor in respect to mismanaged waste of plastics. (Lebreton et al. 2017)

show the average concentration of MP is 1219 ± 2013 particles/m^3 during winter sampling and 101 ± 102 particles/m^3 in spring sampling. The Seine river basin has been reported to receive typically suspended debris of 1937 tons annually (National Statistics 2002).

8 MPs pollution in the North and South American River Basins

If we consider the plastic pollution in the riverine system of the North American Continent, River San Gabriel, River Los Angeles and River Hudson are the most important rivers in this area. Study reveals that MP particles size ranging between 1 and 4.75 mm have been found in these rivers (Moore et al. 2011). The Hudson River basin is enclosed by a petrochemical hub, metal factories, automobile and textile industries and receives industrial pollutants into the water. The constant pressure of increasing population and urbanization makes the river water accessible for plastic and MPs pollution. An earlier study (Miller et al. 2017) reveals that a total of 233 microfibers were recorded in 142 samples indicating that an average of 300 million individual anthropogenic fibers are being discharged along the upper part of the Hudson River per day. St. Lawrence River that flows across the Laurentian Great

Lake (Lake Superior, Huron) is an important hydro-geographic region of North America and serves a vast area of this continent. Sediment samples collected from the 10 sampling sites of the St. Lawrence river show the abundance of MP in every sample (297.97 items per kg) (Gasperi et al. 2014). Amazon, the second largest river of the earth after the Nile, covers about 40% of South America carrying 38,900 tons of plastic particles annually (Lebreton et al. 2017). It represents 20% of the global riverine input into the ocean. The Atoyac river basin of Mexico City, a highly urbanized river system of Central Mexico, carries on average 4500 MP items per kg (Shruti et al. 2019).

9 MPs Pollution in African Riverine Basins

The existence of MPs in the sediment, freshwater system, and the coastal region has been found across Africa. River Imo and Kwa Ibo that are flowing through Nigeria show the distressful situation. River Imo is carrying 21,500 tons and Kwa Ibo is 11,900 tons of plastics annually (Lebreton et al. 2017) (Fig. 6.3) (Table 6.5).

10 Plastic Pollution and MPs Interaction with Biota: Status and Risk Assessment

The interaction of MPs with the aquatic animal is a major concern for scientists nowadays. This has a great impact on the aquatic food chain and therefore ultimately leading to direct and indirect human health impacts. Thus accumulation, transformation and magnification potential of MPs in the freshwater food chain are being urgently monitored throughout the world. The ingestion of MPs was observed within a large variety of biota, from microscopic life forms, for instance, zooplankton (Desforges et al. 2015), small marine organisms such as barnacles, lugworms and mussels (van Cauwenberghe et al. 2015) to large marine organisms such as pelagic fishes (Romeo et al. 2015). A number of field reports have shown the presence of MPs within invertebrates (Su et al. 2016) and in gut contents of several freshwater fishes (Zhang et al. 2017; Jabeen et al. 2017). Marine invertebrates were observed to exhibit different modes in ingestion of MPs, such as filter-feeding animals and deposit-feeding animals can feed MPs directly and thus selectively ingest these (Graham and Thompson 2009). The presence of MPs have been reported in the intestinal lumen, digestive tubule, their epithelial lining and within lysosomal compartments (von Moos et al. 2012) of aquatic organisms. Polystyrene beads (3–9.6 μm) have been reported to translocate haemolymph and persist there for up to 48 days and often can enter the haemocytes also. Many organisms, such as mussels having an open circulatory system, may face serious threats due to the small particle size of MPs as they have enhanced ability to circulate throughout the body

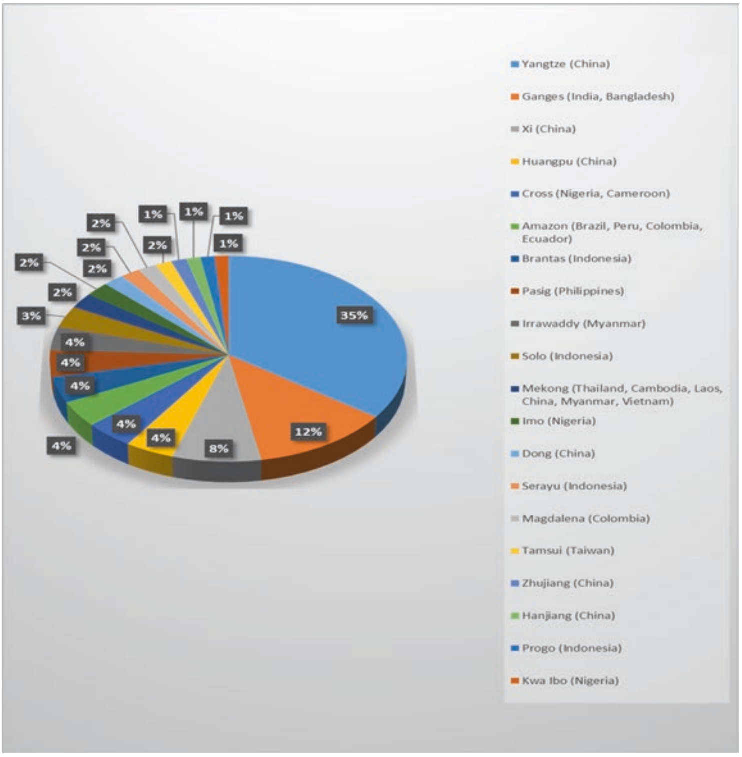

Fig. 6.3 Annual plastic discharge from top 20 rivers into the Ocean (Lebreton et al. 2017)

and have the potential to change the transcriptional profile due to different ways of cellular toxicity and even neurotoxicity (Zhang et al. 2015; Auta et al. 2017; Avio et al. 2015), oxidative stress and damage (Paul-Pont et al. 2016), shift in immune parameters, diminished lysosomal stability (Pittura et al. 2018) and increase in the formation of granulocytoma (von Moos et al. 2012) and total histopathological lesions (Brate et al. 2018). At organismal grade, when mussels were exposed to PS nano plastics (30 nm) chronically, it demonstrated lowered feeding activity and a higher rate of pseudo feces production (Wegner et al. 2012).

Deposition of MPs in the gut and internal tissues may lead to trophic-level transfer of MPs. MPs have been reported to be tropically transferred through mussels to pufferfish and crabs (Farrell and Nelson 2013; Santana et al. 2016). In many cases, it has been reported the presence of MPs foods such as marine fishes and shellfishes even when grown in aquaculture practices. As the MPs are vectors of Persistent organic pollutants (POPs), they have immense potential to transfer these pollutants to human consumer products (Browne et al. 2013; Rochman et al. 2013a; Lusher 2015; Ma et al. 2016). The transfer of adsorbed chemical pollutants, that is, POPs via MPs have been confirmed in annelids, crustaceans, mollusks, fishes and other marine organisms (Tanaka et al. 2013; Oliveira et al. 2013; Batel et al. 2016; Batel

Table 6.5 Top 20 rivers contributing to plastic waste in the aquatic systems throughout the globe. (Lebreton et al. 2017)

Catchment	Location	Input estimate of lower mass (ton/yr)	Input estimate of midpoint mass (ton/yr)	Input estimate of upper mass (ton/yr)	Yearly avg. release (m^3/s)
Yangtze	China	3.10×10^5	3.33×10^5	4.80×10^5	1.58×10^4
Ganges	South Asia	1.05×10^5	1.15×10^5	1.72×10^5	2.08×10^4
Xi	China	6.46×10^4	7.39×10^4	1.14×10^5	5.53×10^3
Huangpu	China	3.35×10^4	4.08×10^4	6.73×10^4	4.04×10^2
Cross	Africa	3.38×10^4	4.03×10^4	6.5×10^4	2.40×10^2
Brantas	Indonesia	3.23×10^4	3.89×10^4	6.37×10^4	8.18×10^2
Amazon	South America	3.22×10^4	3.89×10^4	6.38×10^4	1.40×10^5
Pasig	Philippines	3.21×10^4	3.88×10^4	6.37×10^4	2.07×10^2
Irrawaddy	Myanmar	2.97×10^4	3.53×10^4	5.69×10^4	5.49×10^3
Solo	Indonesia	2.65×10^4	3.25×10^4	5.41×10^4	7.46×10^2
Mekong	South Asia	1.88×10^4	2.28×10^4	3.76×10^4	6.01×10^3
Imo	Nigeria	1.75×10^4	2.15×10^4	3.61×10^4	2.79×10^2
Dong	China	1.57×10^4	1.91×10^4	3.17×10^4	8.54×10^2
Serayu	Indonesia	1.33×10^4	1.71×10^4	2.99×10^4	3.70×10^2
Magdalena	Colombia	1.29×10^4	1.67×10^4	2.95×10^4	5.93×10^3
Tamsui	Taiwan	1.16×10^4	1.47×10^4	2.54×10^4	1.08×10^2
Zhujiang	China	1.09×10^4	1.36×10^4	2.31×10^4	1.33×10^2
Hanjiang	China	1.03×10^4	1.29×10^4	2.19×10^4	7.35×10^2
Progo	Indonesia	9.80×10^4	1.28×10^4	2.29×10^4	2.79×10^2
Kwa Ibo	Nigeria	9.29×10^4	1.19×10^4	2.08×10^4	1.92×10^2

et al. 2018; Guilhermino et al. 2018). Uptake of MPs resulted in augmented bioaccumulation of POPs in lugworms (Besseling et al. 2013), damage in cells and tissues in mussels (von Moos et al. 2012) and endocrine interference in Japanese Medaka (Rochman et al. 2014).

Several studies reported that POPs adsorbed by microplastics cause hepatic stress in fish, amplified depletion of liver glycogen and cellular necrosis (Rochman et al. 2013b). Exposure to HOC via microplastic shows oxidative stress in the lugworm (Browne et al. 2011). When juvenile goby fish was exposed to pyrene sorbed PE MPs, reduced antioxidant functions were observed (Oliveira et al. 2013). When Mediterranean Mussels (*M. galloprovincialis*) was exposed to pyrene-contaminated PS and PE MPs (<100 μm), diminished activity of AchE was found and the frequency of micronuclei formation increased (Pittura et al. 2018).

Further research is needed to explain the consequences of POP-plastic amalgamations on different organisms with varying sizes of plastic. However, benzopyrene is one of the most important priority pollutants studied in genotoxicity and carcinogenicity studies in ecotoxicology (USEPA 2014; Banni et al. 2017; Di et al. 2017; Orbea et al. 2002). Due to its high lipophilicity, benzopyrene has been reported to

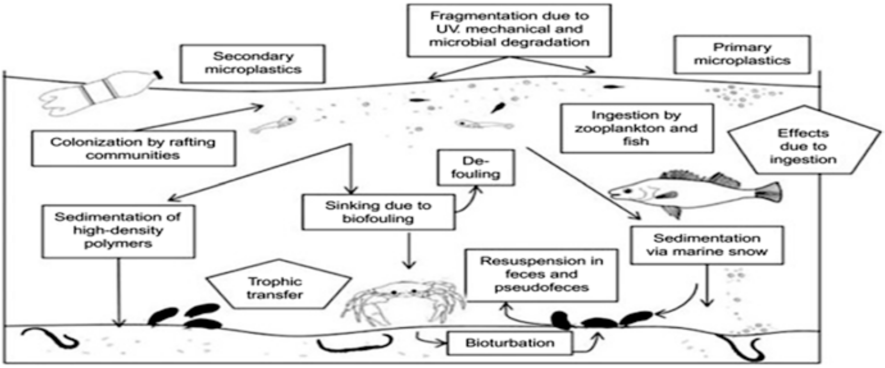

Fig. 6.4 Potential pathways of transportation of microplastics in food web. (Wright et al. 2013)

adsorb sturdily onto MPs and it is transferred via from one trophic level to another (Batel et al. 2016) (Fig. 6.4).

11 Probable Remediation

The emerging threat of plastic and its consequences have been discussed earlier. The substantial growth of plastic polymer in freshwater and marine system is a big headache for the environmentalists and ecologists. The thermal stability, durability, elasticity and nonbiodegrdability of plastic polymer are big challenges to eradicate it from the environment. Over the last 30 years, scientists are looking for new techniques to eradicate plastic from the environment in an ecofriendly manner, and the use of microorganisms give us a beacon of hope in the time of disillusion (Zbyszewski et al. 2014). Microorganisms are kinds of opportunistic and potential organisms that have the ability to adjust even in adverse environmental conditions. Scientific studies reveal that some microorganisms have the outstanding capability in degrading plastic polymer into monomers. Benzopyrene, a PAH with high molecular weight and extreme carcinogenicity, is degraded by the bacteria *S. acidomaniphila* (Aziz et al. 2017). The fungi *Zalerion maritimum* was used to break down polyethylene (PE) into its monomeric form. The results highlight that *Z. maritimum acts as a* potential bioremediation tool in the eradication of PE from the environment (Paco et al. 2017). *Bacillus cereus* has the potential to degrade polyethylene (PE) (Sowmya et al. 2014). Auta et al. (2017) isolated eight bacterial strains from the mangrove ecosystem; of them, two *Bacillus* bacterial strains, namely, *B. cereus* and *B. gotheilii* show the potential effect in the biodegradation of PE, PET, PS and PP. Bacterium *Idonella sakaiensis* can degrade PET and use it as an only energy source for growth (Yoshida et al. 2016). *Pseudomonas sp* and *Bacillus sp* can degrade brominated Polystyrene (Mohan et al. 2016). All these scientific results and reports indicate a new horizon and raise our hope to make a plastic-free world.

12 Concluding Remarks

Globally, the health of most of the rivers are distressed due to anthropogenic influences and plastics have become a new way of river pollution and eventually the most devastating also. In one way or other, it interferes with the aquatic food web and ultimately public health. Empirical studies suggest Asian countries, specifically countries in Southern Asia like China, Philippines, Indonesia, Sri Lanka, Vietnam and Thailand are top producers of mismanaged plastics generation. South Asian rivers like Yangtze, Pearl, Zhugiang, Dong, Xie, Huangpsu, Yujibang of China, Ganges of India, Brantas, Solo, and Serayu of Indonesia, Pasig of Philippines, Irrawaddy of Myanmar and Mekong are worst hit regarding MPs pollution. European rivers like Rhine, Thames, Po of Italy and Seine of France are also deeply affected by the MPs pollution. American rivers Los Angeles, San Gabriel, Hudson, St. Lawrence, Amazon and Atoyac carry a huge amount of MPs in their course. River Imo and Kwa Ibo of Africa are also reported to carry a significant amount of MPs in the aquatic systems. Distributions of the MPs have been noted in almost every possible aquatic habitat from deep sea to the Arctic zone to the Tibet plateau. The global annual input of plastic through the riverine system to the ocean is reflecting the alarming situation of the aquatic system. Improper monitoring of plastic litter makes the water inaccessible to drink and inhabitable for aquatic animals. The ecotoxicological effects of MPs pollution on the freshwater riverine systems indicate the mismanagement of plastic debris by different countries throughout the world. Deposition of MPs in the tissue organization of different aquatic invertebrates and vertebrates have been reported in several studies. Tissue distribution of the MPs have been found in the intestinal epithelium, digestive glands, muscles and body fluids. The secondary effects of MPs due to POPs have emerged as another important way of interruption of food web functioning. These altogether make the ecological imbalances in the aquatic environments. Although several methods, principally based on the application of microbes, are being developed to reduce the MPs load in the aquatic systems, yet these are not sufficient and future orientations towards this field may be enhanced to save our aquatic environments from this distressing situation.

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Chapter 7
Assessment of River Health through Water and Biological Characteristics

Jasmin Rout and Gunanidhi Sahoo

Abstract Water covers 71% of the surface of the earth, out of which the share of rivers and lakes being 0.3% only. Therefore, it is necessary to assess and maintain the health of rivers. Physical, chemical, and biological characteristics are exclusively measured to determine the health of river water. The present study was carried out for 3 years during 2017–2019 to evaluate the ecosystem health of River Mahanadi and its seven tributaries at the nodes by analyzing the physical habitats, chemical characteristics of water, and biological attributes of fishes. River health was calculated from water quality index (WQI), qualitative habitat evaluation index (QHEI), nutrient pollution index (NPI), and biological integrity index (IBI). The major stressors affecting river health have been established through principal component analysis (PCA). Cluster analysis revealed variations in water quality over time, which are typically signs of pollution. We also described a variety of macroinvertebrates to determine their function in determining the quality of river water and pollution levels. The study suggests that most sites of the river have moderate water quality. The water quality of upstream stretches was deteriorated due to anthropogenic activity and intensive agriculture. Nutrient (N and P) enrichment, organic matter contaminants from domestic wastewater disposal, deforestation, and loss of riparian vegetation were identified to be the major stressors. The deterioration of the quality of rivers was also linked to river regulation and channel alteration. The findings of the study could be used as a metric for assessing the ecological health of rivers.

Keywords Environmental impact · Fishes · Macroinvertebrates · Nutrient enrichment · River health · Stressor identification · Water quality

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1 Introduction

Life exists on the blue planet due to the presence of water in it. Although 71% of the earth's surface is covered with water (Annan 2005), freshwater makes up only 2.5% of the earth's water, and less than 0.3% of freshwater is found in rivers, lakes, and the atmosphere. Rivers are the most vital freshwater resource for human beings. In the earlier centuries; social, economic, and political development has been mostly related to the distribution and accessibility of freshwaters contained in riverine systems. Rivers are structurally complex, biologically diverse, and productive due to their dendritic structure, connectivity with adjacent water bodies, and multiple relationships with terrestrial and marine ecosystems (Thorp et al. 2006).

Globally, most of the rivers are experiencing deterioration in water quality (Nash 1993; Loucks et al. 2005; Schwarzenbach et al. 2010), as it flows unidirectionally that eventually connects the upstream and downstream reaches (Thompson and Lake 2010). Therefore, rivers play an important role in assimilating and transporting urban and industrial wastewater and removal of nutrients from agricultural fields and mineral rocks through surface runoff, which pollutes rivers (Vega et al. 1998; Singh et al. 2004). These pollutants have the potential to change the patterns in nutrients (dissolved N and P), suspended sediments, salts, and physical habitat from headwaters to downstream near estuaries, affecting ecological functions of trophic compositions and tolerance species in aquatic biota directly or indirectly over time (Schlossser 1982; Gelwick 1990; Edds 1993). High-suspended sediment concentrations make the water turbid that ultimately leads to a decrease in light penetration, smothering, and scouring of habitats and biota. Both particulate and dissolved sources of nitrogen and phosphorus exist, and high concentrations of dissolved nutrients can cause eutrophication in river systems. High levels of pollution not only jeopardize the use of rivers as a source of potable water for humans (Jiang 2009), but also lead to river depletion (Smith et al. 1999). The most significant inland water supplies for human consumption are rivers; hence data on water quality indices and patterns is essential for successful water management. Water from upstream should be used in such a way that it has no quantitative or qualitative effect on downstream users of water. Subsequently, quality scientific data on the quantity and quality of the waters will be required by the managers of river water. Hence, river water sharing is a significant political issue at most levels. Complete information on river ecosystems is needed for the diagnosis and evaluation of river health, but such an integral approach of river health evaluations is still little understood.

A metacommunity is a set of interconnected communities linked by the dispersal of several potentially interdependent organisms (Leibold et al. 2004; Holyoak et al. 2005; Altermatt 2012). Confluence points of dendritic (river) systems have the maximum diversity and most species abundance (Fernandes et al. 2004; Grant et al. 2007). At confluences, individuals of various species mix, resulting in increased biodiversity at the confluence and downstream. Due to the analytical ease of chemical condition, previous researches on river health have historically concentrated on chemical monitoring. Chemical monitoring may not be sufficient for determining river health, but additional biological and ecological evaluations of aquatic environments are needed for successful management (Karr 1981; Barbour et al. 1996; Yoder and Rankin 1998). Regardless of these details, metadiversity monitoring and assessments in the confluence points, as well as upstream and downstream, should be analyzed.

Macroinvertebrates, as indicators of primary consumers (Plafkin et al. 1989), and fish, as indicators of primary and top consumers (Lang et al. 1989; Lang and Reymond 1995), are the most commonly used biota in river ecosystem health assessments. Since macroinvertebrates have a well-understood taxonomy and are easy to sample, they are chosen as indicators (Smith et al. 1999). Because of their ease of collection and identification in the field, durability in the water, and adaptive response to changes in water chemistry and physical habitat, fishes are one of the best indicators for assessing the health of aquatic environments. Fish taxa are useful for determining long-term effects of environmental change, population growth, obesity, and fish welfare (Munkittrick and Dixon 1989). Models of biological integrity based on fish assemblages have been established regionally (Pyron et al. 2008). Many countries in North America (Karr and Dionne 1991), Europe (Oberdorff and Hughes 1992), South America (Lyons et al. 1995; Soto-Galera et al. 1998), and Africa (Hugueny et al. 1996; Kamdem-Toham and Teugels 1999; Kleynhans 1999) have adopted biological integrity models. Diversity in riverine systems was often linked to local abiotic factors describing patch quality (e.g., pH, temperature, substrate type of riverbed) (Power et al. 1988). It is critical to comprehend the population structure and biodiversity trends in riverine ecosystems. We need a broad understanding of dendritic system ecological processes to preserve this diversity.

The present study was initiated to assess the ecological health of Mahanadi River on the basis of chemical water qualities, the physical habitat, and biological characteristics. We analyzed the contribution of water quality parameters to understand the spatial and temporal variability in water quality, and classified contaminants affecting water quality and their possible sources through multivariate techniques such as Pearson correlation, PCA, and CA. Such studies could pave the way for successful river ecosystem management and restoration in the study area and beyond.

2 Materials and Methods

2.1 Study Area

The Mahanadi River is one of the largest rivers that flows from west to east and eventually empties into the Bay of Bengal. The river arises in a pool at an elevation of 442 m above msl, 6 km from Pharsiya village near the town of Naori in the Raipur district of Chhattisgarh. It is the lifeline of the people of Chhattisgarh and Odisha, which together account for 4.3 percent of the country's total land area. This

Fig. 7.1 Sampling locations along river Mahanadi and her tributaries (Source: Google earth)

study was carried out on 851 km long stretch of the river and her tributaries encompassing 141,589 km^2 basin area between 80° 30′ and 84° 50′ East and 19° 20′ and 23° 35′ North. The three major tributaries of River Mahanadi are the Seonath (or Shivnath), Ib, and Tel, the other important tributaries being Hasdeo, Mand, Jonk, and Ong.

Sheonath, the first main tributary of Mahanadi, is connected to Mahanadi on the right by Khargahni in Bilaspur district and is named Sheonath River node (Fig. 7.1). River Jonk joins Mahanadi on the northern side at Seorinarayan and formed the Jonk River node. The Hasdeo River enters from the left about 17 km later, near Mahuadih, the confluence point being the Hasdeo River node. On the left of Chandrapur, the Mand River joins Mahanadi farther down the road (Mand River node). Mahanadi leaves Chhattisgarh and enters Odisha after another 28 km near Hirakud. The river lb joins from the left, forming the Ib River node near Bagra. The Ong River enters the Mahanadi from the right about 11 km upstream of Sonepur district (Ong River node). The river turns towards the southeast in the vicinity of the Sonepur district and is connected by the second largest tributary, the Tel, on its right (Tel River node). The study was conducted on monthly basis at the seven confluence points/nodes of Mahanadi over three consecutive years (2017–2019). Sampling was done at the nodes (S0 for Shivnath), 1 km upstream (S1) and 1 km downstream (S2) of the nodes. The names of the rivers are represented by their first letter.

2.2 Meta Diversity Analysis

Macroinvertebrate samples were taken across the sampling points to facilitate recovery of microhabitat-specific invertebrates (Mason, 2002; Ziglio et al. 2006). We took a kick sample of 3 min as described in Abel (1996) from each sampling point. A pond net (250 μm mesh) was used to capture the macroinvertebrates and then washed to remove gravel, sand, large detritus, etc. Selected animals were stored in 70% alcohol for family-level identification.

Fish sampling was conducted twice during the breeding season (May–September). With the assistance of locally recruited experienced fishermen, experimental fishing was conducted in all sampling locations. Gillnets (mesh 2.5 × 2.5 cm; 3 × 3 cm; 7 × 7 cm; length × breadth = 75 × 1.3 m; 50 × 1 m), cast nets (mesh 0.6 × 0.6 cm), drag nets (mesh 0.7 × 0.7 mm, L × B = 80 × 2.5 m with varying mesh sizes) and fry collecting nets (nets made of nylon mosquito nets that are tied at both ends with bamboo) were used to catch the fishes. All the species were described in situ using the classification system of Nelson (2006), scientific names were checked using http://www.fishbase.org before being released. Unidentified specimens, on the other hand, were stored in 10% formaldehyde for further identification. Anomalous external properties such as deformities (D), erosion (E), lesion (L), and tumor (T) were tested for all the fishes sampled following Sanders et al. 1999.

2.3 Analysis of Water Quality Parameters

The discharge of domestic sewage, industrial and biomedical wastes into river Mahánadi through her tributaries (e.g., Shivnath River, Hasdeo River, and Ib River) has raised concerns and posed significant threats to the health of the aquatic fauna and nearby human inhabitants. Monthly water samples were collected from each study site during 2017–2019 to record 15 water variables such as pH, temperature, total suspended solids (TSS), electrical conductivity (EC), total dissolved solids (TDS), dissolved oxygen (DO), biological oxygen demand (BOD), chemical oxygen demand (COD), turbidity (TURB), total nitrogen (TN), total phosphorus (TP), ammonia nitrogen (NH_4-N), nitrate nitrogen (NO_3-N), total phosphate (Po_4-P), and chlorophyll-a (CHL-a). pH, TSS, and EC were measured instantly with an Ecotester-10 (Eutech Instruments). Water temperature was measured by a thermometer. TDS, DO, BOD, COD, TURB, TN, TP, NH_4-N, NO_3-N, PO_4-P, and CHL-a were analyzed in the laboratory following standard methods (Prepas and Rigler 1982; APHA 1985; Rodier et al. 2009).

2.4 Water Quality Index (WQI)

The National Sanitation Foundation (NSF) has recommended the Water Quality Index (WQI) to compare the relative quality of different water bodies (Horton 1965). The WQI was developed through a combination of nine factors [pH, temperature, turbidity, TSS, DO, BOD, nitrates, TP, and FC (Fecal Coliform)] to determine the water quality. The weightage of the study was slightly altered as eight parameters were used (without fecal coliform) for the purpose. The modified total weight score stayed at 1 despite the modifications. The weight score modification was proportional to its original value. The weight score (W_i) was multiplied by the subindex value (L_i) and then summed to obtain WQI.

$WQI = \Sigma W^n_i \times L_i$ where, i = 0. where, W_i: The weight score, L_i: The subindex value.

The range of WQI scores was used to classify the waters of the sampling area (Table 7.1).

2.5 Nutrient Pollution Index (NPI)

The chemical health of river waters was assessed using the nutrient pollution index (NPI) (Dodds et al. 1998; Lee and An 2009), M_1: total nitrogen (TN, mg/L), M_2: total phosphorus (TP, μg/L), M_3: TN: TP ratio, M_4: BOD (mg/L), M_5: total suspended solids (TSS, mg/L), M_6: electrical conductivity ($\mu S \cdot cm^{-1}$), and M_7: Chlorophyll (mg/L) were the seven metrics graded under the categories of the nutrient regime, organic matter, ionic contents and solids, and primary production indicator. Each metric had received a score of 5, 3, or 1 point. In order to evaluate the water quality, the data from the above seven matrices were added and categorizing the water as excellent (Ex: 31–35), good (G: 25–29), fair (F: 19–23), poor (P: 13–17), and very poor (VP: 7–11).

Table 7.1 WQI classification criteria

Sl. No.	NSF-WQI Score	Criteria
1	0–25	Very bad
2	26–50	Bad
3	51–70	Medium
4	71–90	Good
5	91–100	Excellent

2.6 Qualitative Habitat Index (QHEI) and Assessment of Physical Habitats

The health of physical habitats at various sampling locations was assessed with the QHEI index. Six metric models (M_1: epifaunal substrate/available cover, M_2: pool substrate characterization, M_3: channel flow status, M_4: the existence of small-scale dams, M_5: channel alteration, and M_6: sediment deposition) were extracted from the initial 11-metric QHEI models (Plafkin et al. 1989) for our analyses. Health of habitats was assessed by combining results obtained from the six parameters and divided into four categories: excellent (Ex: score 96–120), good (G: 66–80), fair (F: 36–60), and poor (P: 6–30).

2.7 Index of Biological Integrity (IBI)

The biological integrity was evaluated following the IBI model (Karr 1981; Karr et al. 1986). Ecological characteristics like species richness and composition, trophic composition, and fish abundance with health status were the three main categories of these metrics. IBI included M_1: total number of native species, M_2: number of riffle-benthic species, M_3: number of sensitive species, M_4: proportion of individuals as tolerant species, M_5: proportion of individuals as omnivore species, M_6: proportion of individuals as insectivore species, M_7: total number of native individuals, and M_8: percent of individuals with anomalies were the individual metrics. The highest species richness was used to rank M1, M2, M3, and M7 (Yoder and Rankin 1998). Each metric was assessed as 5, 3, or 1 and health conditions at the community level were evaluated following Barbour et al. (1999). Excellent (Ex: 36–40), good (G: 28–34), fair (F: 20–26), poor (P: 14–18), and very poor (VP: 8–13) were the five categories of IBI ratings.

In the analysis of nutrient pollution index (NPI), qualitative habitat evaluation index (QHEI), and biological river health assessment index (IBI), the average of all the seven node points was presented as node. The averages of seven upstream and downstream points were denoted as group I and group II, respectively. The node regions were considered as the reference sites and compared to the group I and II regions to determine the similarity and dissimilarity with respect to the pollution level, habitat assessment, and biological integrity.

2.8 Statistical Analyses

Multivariate tests such as Pearson's correlation, PCA, and CA were applied to the spatially and temporally generated variables. The relationship between water quality variables was assessed with a Pearson correlation. Kaiser–Meyer–Olkin (KMO) was used to check the data for normality and the effective factorization of the calculated variables was tested using Bartlett's Sphericity tests. PCA was used to determine the association between the water variables studied using reduced structured data sets (Mahlknecht et al. 2004; Srivastava and Ramanathan 2008). The variance in square loadings was maximized with the varimax rotation. The relationship between macroinvertebrate communities and environmental parameters was investigated using canonical correspondence analysis, which led to the identification of corresponding trends in macroinvertebrate composition. Cluster analysis was used on the normalized data using Ward's procedure and squared Euclidean distances as a measure of similarity to examine similarity and dissimilarity of the composition among the sampling locations (temporal and spatial variability) (Massart and Kaufman 1983; Singh et al. 2005; Zhao et al. 2011). Ward's approach calculates the rise in squared error as the distance between two clusters gets closer. This is the most widely used approach to classifying groups more precisely. The result is shown through a dendrogram showing the clusters and their proximity, with the original data's dimensionality reduced (Forina et al. 2002). CA was also performed in the analysis of macroinvertebrate communities to produce a similarity matrix with Bray–Curtis indices based on an abundance matrix of species transformed by square root to generate a dendrogram. Microsoft Excel 2007 and IBM SPSS 20 were used to perform all statistical calculations in this analysis.

Table 7.2 Physicochemical characteristics of water in and around the seven confluence points in Mahanadi River and its tributaries

	pH	Temp	TSS	EC	TDS	DO	BOD	COD	TURB	TN	TP	NH_4-N	NO_3-N	PO_4-P	CHL-a
S1	8.26	26	417.76	354	148	8.6	4	21.15	13	3.3	0.54	0.74	0.47	0.27	5.82
S0	8.13	26	452.69	168	127	7.8	3.1	17.86	13	1.8	0.31	0.45	0.37	0.15	2.86
S2	8.09	28	58.9	158	114	3.7	4.3	19.89	10	1.5	0.28	0.34	0.38	0.11	6.07
H1	8.2	29	284.11	313	138	8.8	4.2	21.57	13	2.64	0.44	0.91	0.44	0.23	7.35
H0	8.14	28	208.47	103	118	3.5	1.59	21.34	11	1.84	0.21	0.67	0.26	0.12	3.11
H2	8.03	29	72.7	141	113	5.1	3	22.62	10	1.12	0.24	0.3	0.38	0.09	6.28
J1	7.6	29	346.32	146	111	7.9	0.8	19.58	11	2.48	0.21	0.28	0.46	0.06	1.62
J0	7.42	29	207.23	160	89	9.2	0.4	20.72	11	1.12	0.26	0.15	0.32	0.01	0.76
J2	7.14	29	40.25	124	57	8.3	1.98	18.85	8	1.2	0.19	0.09	0.37	0.03	1.13
M1	7.33	29	87.74	161	92	8.2	4	29.08	9	1.56	0.22	0.08	0.37	0.09	6.82
M0	7.21	22	58.7	167	116	8.5	0.4	20.56	8	1.98	0.21	0.08	0.47	0.05	0.57
M2	7.69	26	250.1	186	82	8.9	0.4	25.65	10	1.26	0.11	0.09	0.38	0.07	1.42
I1	8.22	20	158.42	287	155	8.5	3.85	17.47	13	2.84	0.5	0.54	0.44	0.06	7.60
I0	8.1	29	42.88	105	126	3.8	2.23	17.61	8	1.68	0.11	0.4	0.33	0.09	0.82
I2	8.01	21	36.79	125	109	8.9	2.08	23.74	8	1.9	0.23	0.11	0.36	0.02	0.72
O1	7.5	28	58.9	134	91	6.7	1.66	16.1	9	1.84	0.3	0.09	0.42	0.04	0.85
O0	7.23	26	63.35	146	96	7.9	0.84	18.98	9	2.4	0.38	0.17	0.38	0.09	1.44
O2	7.38	22	60.48	123	70	8.7	1.46	21.89	8	2.32	0.22	0.09	0.29	0.08	0.93
T1	7.9	29	63.06	190	87	8.1	2.05	19.6	8	1.14	0.37	0.11	0.37	0.09	0.78
T0	7.27	25	82.08	173	115	8.1	1.26	17.44	9	2.28	0.15	0.14	0.33	0.04	1.31
T2	7.48	27	86.95	168	70	7.5	1.88	28.8	8	1.4	0.31	0.05	0.3	0.08	0.67

3 Results and Discussion

3.1 Assessment of Water Quality

In the study region, the water was alkaline (pH: 7.23 to 8.36): being higher at S1 (Shivnath River upstream) and lower at J2 (Jonk River downstream) (Table 7.2). The physical, chemical, and biochemical properties of water are influenced by the water temperature. In the present study, temperature ranged between 10 °C and 32 °C, with comparatively higher temperatures at H1, H2, J1, J0, J2, M1, I0, T1, and lower at I1. Generally, water temperature is inversely proportional to water depth. The TSS ranged between 10.8 and 1328.54 mg/L. The average TSS peaked at the confluence point when Shivnath meet to Mahanadi and decreased downstream. Most of the upstream regions recorded higher TSS values than nodes and downstream indicating the urban wastewater discharges. The EC level ranged between 92 and 1025 μS/cm^{-1} which was high at Shivnath River (354 μS/cm^{-1}, Table 7.2). TDS, a measure of inorganic salts and trace quantities of organic matter in water, was varied between 62 and 484 μS/cm^{-1} with highest at Shivnath River (Table 7.2).

The dissolved oxygen content of water benefits both the flora and fauna of aquatic ecosystems, making it a significant parameter in water quality research. It is often used for water quality assessment (Sanchez et al. 2006; Qadir et al. 2008; Barakat et al. 2016; Nayak et al. 2017; Ganguly et al. 2018). DO concentration ranged between 2.7 to 9.4 mg/L. The Hasdeo node point recorded lower values (average 3.5 mg/L) due to local domestic wastewater, increase in microbial activity, as well as increase in other pollutants. Higher DO at Jonk River node (average 9.2 mg/L) was the result of decreased pollution load and consequent reduction in microbial activity. Introduction of oxygen-intensive materials (organic or inorganic) decreases the DO content, posing a serious threat to aquatic organisms, especially fish and other higher taxa. Accumulation of agricultural, industrial, and household effluents containing oxidized organic substances and resultant biodegradation reduces dissolved oxygen content (Adeleye and Adebiyi 2003). The organic pollution load is measured through BOD and COD (Galal-Gorchev et al. 1993). BOD varied between 0.2 and 12 mg/L indicating the water to be ranked as excellent to pollute as per WHO standards. COD is commonly used to determine the amount of wastes in water (Kazi et al. 2009). COD levels ranged from 4.26 to 87.42 mg/L, highest concentration being at the Mand River (average 29.08 mg/L).

Higher levels of turbidity indicate higher concentrations of particulate matter deposited in water due to anthropogenic inputs. The turbidity concentration ranged from 0.6 to 41 NTU, suggesting the river water to be moderate to heavily polluted. TN concentration ranged between 0.4 and 10.2 mg/L, whereas TP varied between 0.06 and 2.64 mg/L. Maximum TP and TN concentrations were recorded at Shivnath River point (average 3.3 and 0.54 mg/L for TN and TP, respectively). NH_4^+ is a good indicator of organic contamination that results from the decomposition of domestic, agricultural, and industrial wastes, as well as nitrogen-containing organic materials and gas exchange between water and the atmosphere (Chapman and Kimstach

1996). The NH_4^+ content varied between 0 and 2.32 mg/L. Maximum NH_4^+ was recorded at Hasdeo River point (average 0.91 mg/L) and minimum at downstream of Tel River (average 0.05 mg/L). Nitrates (NO_3^-) are formed in water due to bacterial action and oxidation of ammonia. High nitrates in water indicated organic pollution. Presence of nitrate indicates the oxidation of nitrogenous organic matter (Nayak et al. 2017). Considerably lower amounts of nitrate were detected in the study area. Increased nitrate concentration in water bodies greatly affects the aquatic organisms. NO_3^- was 0.14–1.53 mg/L and was considerably safer as per WHO (50 mg/L) standards. PO_4 was 0.01–2.81 mg/L with highest at Shivnath River point (average 0.27 mg/L) and lowest at Jonk River node (average 0.01 mg/L). Chlorophyll concentration (0.42–14.86) was highest at Ib River point (average 7.60 mg/L) and lowest at Mand River node (average 0.57 mg/L, Table 7.2).

pH was significantly correlated with TDS, BOD, turbidity, PO_4 and, CHL-a; but negatively correlated with DO and COD (Table 7.3). Temperature was found to have a negative correlation with EC, TDS, DO, TN, TP, and NO_3. DO was inversely proportional to temperature as dissolution of oxygen in river water decreases with an increase in temperature that enhanced the activities of microorganisms that necessitated more amounts of oxygen for metabolism and degradation of organic wastes (Kumarin et al. 2013). Significant positive correlations of TSS with EC, TDS, $NH_{4,}$ and PO_4 indicate the presence of colloidal substances (clay, silt, inorganic, and organic matters). EC had a significant positive correlation with TDS, BOD, turbidity, TN, NH_4, NO_3, PO_4, and CHL-a indicating moderate pollution. TDS was negatively correlated to DO and COD; but positively correlated with turbidity, TN, TP, NO_3, $PO_{4,}$ and CHL-a. Lower DO values decrease nitrification rates (Ruiz et al. 2006), which resulted in its positive correlation with NO_3, but reversed with NH_4. DO was negatively linked to BOD, NH_4, $PO_{4,}$ and CHL-a. BOD was significantly correlated with turbidity, TP, $NH_{4,}$ and PO_4 suggesting the influence of wastewater effluents. COD was negatively linked to TN, TP, $NH_{4,}$ and NO_3 and positive with temperature, EC, DO, PO_4, and CHL-a. Little or no correlation was observed between COD and BOD signifying the major part of organic material in the river water not to be biodegradable. COD may be linked to the leaching and transport of contaminants such as natural and domestic sewage, agricultural and industrial pollutants. Turbidity was significantly correlated with TN, TP, PO_4, and CHL-a. Total nitrogen was found to have a strong positive association with TP, NH_4, NO_3, and PO_4. Total phosphorus was positively correlated with pH, TSS, EC, TDS, DO, BOD, turbidity, TN, NH_4, NO_3, PO_4, and CHL-a indicating its source to be natural and anthropogenic. Both CHL-a and NH_4 were positively correlated.

The KMO index compares the relationship between variables and the skewed correlations. PCA is justified when the KMO index is close to one but is irrelevant when close to zero. The KMO values below 0.5 are detrimental, while those between 0.5 and 0.7 are acceptable, and those above 0.7 are excellent (Nnorom et al. 2019; Tripathi and Singal 2019; Patil et al. 2020). The KMO was 0.568 in our study. The null hypothesis that the intercorrelation matrix was derived from a population of non-correlated variables was tested using Bartlett's sphericity test (Shrestha and Kazama 2007). At a significance level of 0.05, the null hypothesis was dismissed,

Table 7.3 Correlation matrix of water quality parameters (Pearson correlation coefficients (r)) in the study area

	pH	Temp	TSS	EC	TDS	DO	BOD	COD	TURB	TN	TP	NH_4	NO_3	PO_4	CHL-a
pH	1														
Temp	0.003	1													
TSS	0.432	0.109	1												
EC	0.424	−0.156	**0.525***	1											
TDS	**0.715*****	−0.227	**0.462***	**0.579****	1										
DO	−0.377	−0.384	0.219	0.415	−0.174	1									
BOD	**0.622****	0.089	0.176	**0.522***	**0.505***	−0.204	1								
COD	−0.119	0.053	−0.05	0.003	−0.344	0.173	0.074	1							
TURB	**0.607****	0.023	0.833	**0.676*****	**0.699*****	0.082	**0.46***	−0.19	1						
TN	0.239	**−0.446***	0.412	0.61**	**0.633****	0.253	0.253	−0.276	**0.498***	1					
TP	0.418	−0.152	0.353	0.787	**0.473***	0.26	**0.541***	−0.138	**0.583****	**0.565****	1				
NH_4	0.767	0.103	**0.598****	**0.609****	0.781	−0.238	**0.58****	−0.219	0.798	**0.558****	**0.537***	1			
NO_3	0.135	−0.116	0.324	0.594**	**0.466***	0.326	0.245	−0.235	0.383	**0.474***	**0.464***	0.25	1		
PO_4	**0.59****	0.185	**0.604****	**0.676*****	**0.532***	−0.084	**0.615****	0.062	**0.606****	**0.502***	**0.587****	0.792	0.288	1	
CHL-a	0.544*	0.066	**0.28**	**0.598****	**0.593****	−0.143	0.848	0.133	**0.646****	0.331	**0.522***	**0.645****	0.358	**0.571****	1

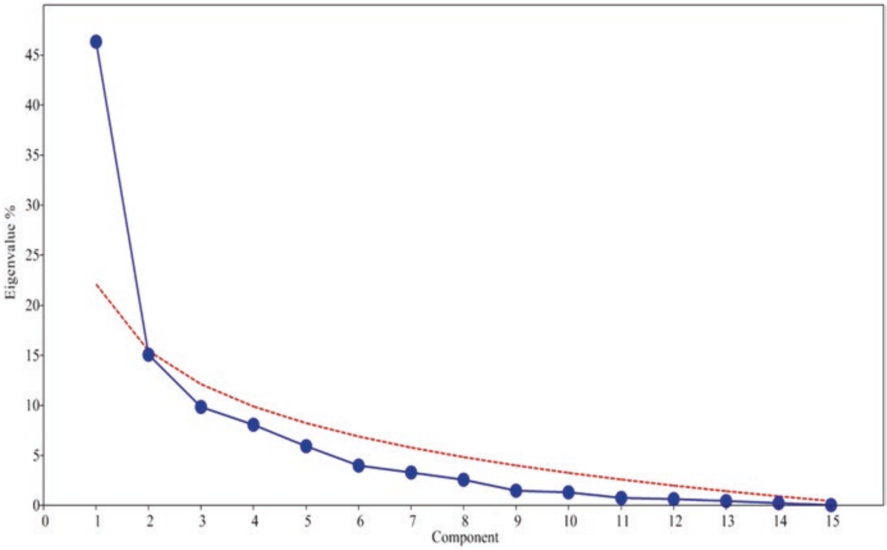

Fig. 7.2 Scree plot of the eigenvalues developed from PCA in the analysis of water parameters of Mahanadi River shed

Table 7.4 Principal component factor loading values in Mahanadi River and her tributaries (in and around the confluence points)

Variables	PC1	PC2	Variables	PC1	PC2
pH	0.727	−0.436	TN	0.672	0.468
Temp	−0.061	−0.622	TP	0.754	0.265
TSS	0.661	0.1	NH_4	0.887	−0.273
EC	0.835	0.346	NO_3	0.529	0.459
TDS	0.83	−0.04	PO_4	0.812	−0.185
DO	0.025	0.862	CHL-a	0.762	−0.258
BOD	0.7	−0.354	Eigenvalues	6.951	2.261
COD	−0.163	−0.086	% of variance	46.343	15.074
TURB	0.867	0.001	Cumulative%	46.343	61.417

but a value of zero in our study was small enough to reject the null hypothesis. The distribution of biological components and selected water quality parameters from upstream to downstream regions was studied using principal component analysis (McCune & Mefford 1999). Without altering information, the PCA reduces complex multivariate datasets to a small and manageable number of factors (Jolliffe 2011; Awomeso et al. 2020) and identifies the features of water quality variables of all sampling points by revealing relationships between them. The number of principal components of the basic data structure was determined using scree plot graphs (Liun et al. 2003). The scree plot (Fig. 7.2) displayed the sorted eigenvalues as a function of the PC number, from large to small. The graph reveals a clear shift in slope following the second eigenvalue suggesting the retention of two components. Table 7.4 shows the loadings of two retained PCs. The results of the PCA showed that the first two axes collectively represent 61.41% of the total variation (Table 7.4). Out of this, 46.34% is explained by PC1 with pH, TSS, EC, TDS, BOD, turbidity, TN, TP, NH_4, PO_4, and CHL-a mainly responsible for the observed variation. This may be due to the physical and chemical properties of water, as well as the river's natural weathering. pH, TSS, and turbidity were mainly generated by runoff from point sources of contamination such as agricultural fields and domestic areas, which had high concentrations of solids and wastes (Gazzaz et al. 2012).

Municipal and industrial point sources, non-point agricultural sources, livestock activities, and domestic sources may all be linked to variables like BOD, TP, EC, and NH_4 (Simeonov et al. 2003). The highest DO value attributable to biogenic and anthropogenic sources is clarified by PC2 (15.07, Table 7.4). Turbidity and TSS had a positive and significant impact on the samples from the Shivnath River node. The observed values at Jonk River could be linked to the effects of livestock breeding activities in the region. Physical parameters like COD, which was closely linked to the majority of the sampling points, also had a negative impact on PC1 (Fig. 7.3). CA was used to organize all sampling points into spatially and temporally related groups based on the similarity of water quality parameters. Figure 7.4 shows the dendrogram achieved by Ward's method in the Mahanadi River and its tributaries. All the 21 study sites were grouped into five clusters by spatial CA, each having similar water characteristics. Cluster 1 included the majority (13) of sampling points (S2, M0, T0, O1, O0, H2, M1, T2, T1, J2, O2, I0, and I2). Cluster 2 included only two (S0 and J1) sites. Both these clusters are associated with moderate levels of pollution. Only one sampling location (S1), downstream of the urban wastewater dump of Bilaspur city, was covered under cluster 3. Cluster 4 included three points (J0,

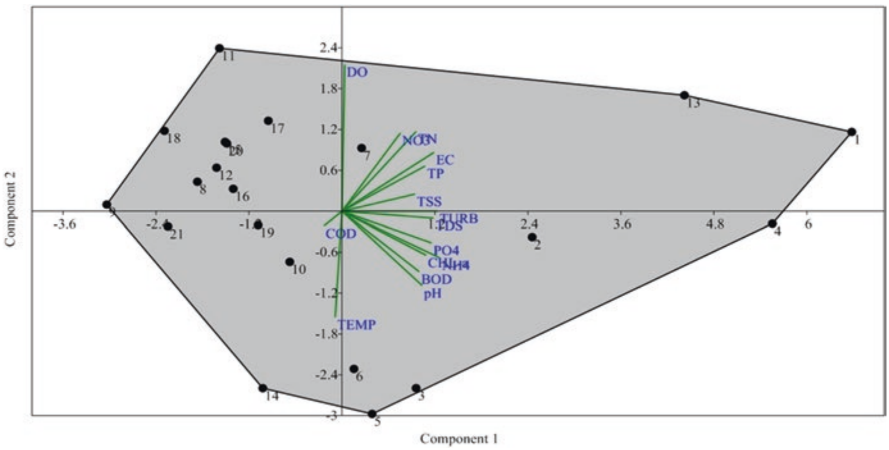

Fig. 7.3 Principal components analysis for the variables of Mahanadi River and her tributaries. Physicochemical parameters (green lines) are pH, temp, TSS, EC, TDS, DO, BOD, COD, TURB, TN, TP, NH_4, NO_3, PO_4, and CHL-a. Points denote for sampling sites

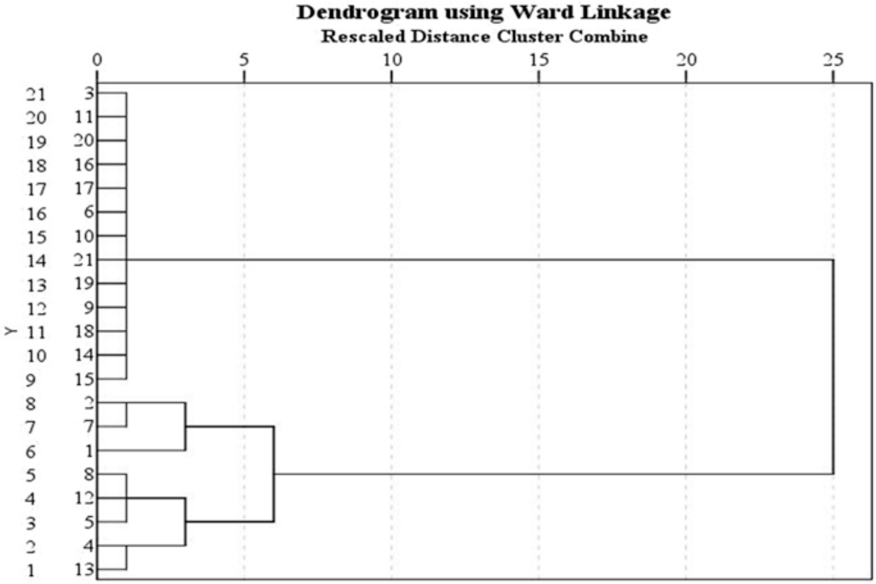

Fig. 7.4 Dendrogram for complete sampling points in Mahanadi River and her tributaries based on hierarchical clustering (Ward's method)

M2, and H0) while cluster 5 contained two (H1 and I1). Clusters 3, 4, and 5 correspond to high levels of pollution when compared to clusters 1 and 2. This is due to urban wastewater discharge, soil leaching through intensive agriculture and livestock activities.

3.2 Water Quality Index (WQI)

The WQI values express water qualities by combining measurements of various water quality variables and presenting the information in a logical and simplified manner (Semiromi et al. 2011). The current study offers details on the Mahanadi River's overall water quality in and around tributaries and confluence points. The WQI of River Mahanadi was 47.68 (Table 7.5) which indicated the water to be polluted. Similar results were obtained by Nayak et al. (2017) in their surface water quality evaluation studies. In general, the sampling sites did not record considerable improvement in WQI during the study period. In the current research, pH, TSS, DO, and BOD all had a major impact on the river's WQI.

Table 7.5 Calculation of WQI from Mahanadi River and her tributaries at confluence points

S. No.	Parameter	Test result	Q value	Weighting factor	Subtotal
1	pH	7.73	85	0.12	10.20
2	Temp	26.52	12	0.11	1.32
3	TSS	149.42	18	0.16	2.88
4	DO	7.46	7	0.18	1.26
5	BOD	2.16	78	0.12	9.36
6	TURB	9.85	77	0.09	6.93
7	TP	0.27	50	0.11	5.5
8	NO_3-N	0.37	93	0.11	10.23
	FC	0	98	NM	NM
TOTAL = 1					47.68
WQI					47.68

Table 7.6 Nutrient pollution index (NPI) of Mahanadi River (node point; upstream and downstream to the node point)

Category	Metrics (M)	Scoring criteria			Mahanadi River watershed		
		5	3	1	Node	Group I	Group II
Nutrient regime	M_1: Total nitrogen (mg/L)	<1.5	1.5–3.0	>3	1.87 (3)	2.25 (3)	1.52 (3)
	M_2: Total phosphorus(mg/L)	<30	30–100	>100	23 (5)	36 (3)	22.57 (5)
	M_3: TN:TP ratio	>50	20–50	<20	42.97 (3)	48.28(3)	72.89 (5)
Organic matter	M_4: Biological oxygen demand (mg/L)	<1	1–2.5	>2.5	1.40 (3)	2.93 (1)	2.15 (3)
Ionic contents and solids	M_5: Total suspended solid (mg/L)	<4	4–10	>10	15.93 (1)	20.23 (1)	8.65 (3)
	M_6: Electrical conductivity (μS/ cm)	<180	180–300	>300	146 (5)	226.42 (3)	146.42 (5)
Primary production indicator	M_7: Chlorophyll (mg/L)	<3	3–10	>10	4.28 (3)	5.36 (3)	1.20 (5)
Final NPI scores					23 (F)	17 (P)	29 (G)

3.3 *Chemical Health Evaluation Based on Nutrient Pollution Index (NPI)*

The nutrient pollution index was used by several workers to measure the nutrient and organic pollution in water bodies (Dodds et al. 1998; Lee and An 2009; Kim and An 2015; Atique and An 2018). Depending on the amount of organic and nutrient contaminants in the water, the index classifies it as oligotrophic, mesotrophic, or eutrophic. The NPI in the present study included seven metrics (M_1–M_7) and were divided into four groups such as nutrient composition (TN, TP, and TN: TP), organic matter (BOD), ionic contents and solids (TSS and EC), and Primary Production Indicator (chlorophyll) (Table 7.6). Total nitrogen (TN) and total phosphorus (TP), as well as the N:P mass ratio, were chosen as NPI model variables because they are considered to be important factors in controlling eutrophication and water quality (An and Jones 2000; Forsberg and Ryding 1980; Sakamoto 1966; Smith 1983; Kim and An 2015). TN metrics were graded as oligotrophic (<1.5 mg/L), mesotrophic (1.5–3 mg/L), and eutrophic (>3 mg/L) based on chemical parameters. This criterion was similar to that of Antique and An (2018) and Kim and An (2015). The TN concentrations at all mesotrophic sites ranged from 1.5 to 3.0 mg/L, indicating the absence of significant spatial variance in the model ranking. Group I showed the highest mean TN value (2.25 mg/L). Node points and downstream regions (group II) were categorized as oligotrophic (23 and 22.57) on the basis of TP levels (<30 μg/L—oligotrophic, 30–100 μg/L—mesotrophic, >100 μg/L—eutrophic). Group I had the highest mean (36) and was classified as mesotrophic, suggesting significant phosphorus enrichment in upstream areas. TN:TP ratio revealed that node group and the group I had a moderate 20–50 ratio, while group II had a high >50 ratio. According to Fujimoto and Sudo (1997) and Seppala et al. (1999), the N:P ratio in ambient waters was used as an indirect measure of nutrient limitation in algal biomass and primary productivity. Contaminated and eutrophic water bodies have a lower N:P ratio (Dodds et al. 1998). The average chlorophyll concentration indicated that group I was oligotrophic (CHL-a < 3 mg/L). Node points and upstream areas, on the other hand, were mesotrophic. BOD is used as a measure of organic load and the metric values for BOD were 1.40 in the node points and 2.93 and 2.15 in the upstream and downstream regions, respectively. The Nutrient Pollution Index (NPI) revealed significant spatial variations among the Mahanadi watershed regions. NPI in the node regions (23) indicated the chemical health to be fair (F). Group II regions had an index value of 29 indicating "good condition (G)" in the downstream. However, NPI (17) in the upstream regions (group I) suggested a "poor (P) condition." The effluents from point/nonpoint sources of municipal wastewater disposal and urban runoff were the reasons for the deteriorated chemical health in the upstream regions in the Mahanadi watershed.

3.4 Physical Habitat Health Assessment Based on Qualitative Habitat Evaluation Index (QHEI)

The QHEI was developed to understand the river habitats and ecological integrity better. The physical habitat health assessment in the Mahanadi River watershed was based upon six metric models. Analysis of QHEI showed that Mahanadi River to be under the good category (G) (Table 7.7). The QHEI value at node points (72.2) and downstream regions (71.5) indicated a Good (G) category. The upstream region (62.4) was classified as Fair to Good (F-G) category (Table 7.7). The upstream to downstream spatial changes were not very significant. The physical habitat health of upstream regions was mainly influenced by anthropogenic activities. Poor epifaunal substrate/available cover (M_1) and pool substrate characterization (M_2) were the main causes for health deterioration in the upstream of Mahanadi River.

Table 7.7 Qualitative habitat evaluation index (QHEI) for the assessment of physical health of Mahanadi River

Metrics	Study areas		
	Node	Group I	Group II
M_1: Epifaunal substrate/available cover	15.8	5.1	11.9
M_2: Pool substrate characterization	15.4	8.6	11.8
M_3: Channel flow status	11.4	12.6	13.7
M_4: Existence of small-scale dams	4.4	12.2	12.5
M_5: Channel alteration	11.9	10.1	9.2
M_6: Sediment deposition	13.3	13.8	12.4
Final QHEI scores	72.2 (G)	62.4 (F-G)	71.5 (G)

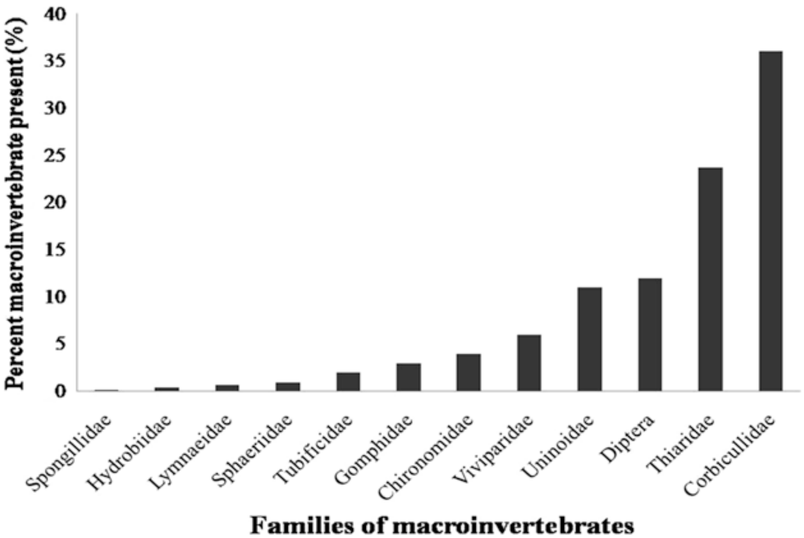

Fig. 7.5 Diversity of macroinvertebrates in and around the confluence points of River Mahanadi and her tributaries

3.5 Macroinvertebrates

Macroinvertebrates are dynamic organisms found in all aquatic habitats. The biomonitoring approach using macroinvertebrates provides crucial information about the characteristics of water, contamination level, and future management options (Hughes and Peck 2008). A total of 426 taxa were collected during the study. The node regions with moderate water current velocity had the highest abundance and diversity. Several gastropods (215 taxa) and bivalve (155 taxa) families have been recorded. Thirteen species of dragonflies (larvae) and midges (*Chironomous*) were also recorded. The invertebrate larvae (Chironomidae, Diptera, and Gastropoda) were also recorded in winter samplings. Family Corbiculidae, with 36% of the total macroinvertebrates, was the most abundant group followed by Thiaridae (23.7%), Diptera (12%), Unionoidae (11%), Viviparidae (6%), and other insect families (Fig. 7.5).

According to Macneil et al. (2002), macroinvertebrate families are a diverse community that is vulnerable to contamination, making them ideal for water quality assessment. The biotic index value was around 8 in the present study which indicated moderate pollution in Mahanadi River shed. For water-quality assessment, Czerniawska-Kusza (2005) employed a number of biological indices focused on macroinvertebrates. Duran (2006) tested the water quality of the Behzat Stream in Turkey using benthic macroinvertebrates. Most of the tolerant species were found in all the 21 sampling sites. The presence of a variety of Gastropods and Dipteran larvae suggested the water quality was moderate. Pollution-tolerant taxa such as Odonata and Trichoptera, as well as pollution-sensitive taxa such as Ephemeroptera, were found in the study locations. In each site, we identified the relationship between macroinvertebrates and the environment. Macroinvertebrate communities were associated with COD, BOD, CHL-a, PO_4, NH_4, turbidity, and TDS (Fig. 7.6).

The length of the arrow is proportional to the overall rate of change in the corresponding variable's value. Diptera and Corbicullidae were among the taxa with negative scores on axis 2, while Chironomidae and Sphaeriidae had high positive scores on axis 2 (Fig. 7.6). Axis 2 had a high score for Chironomidae, but there is a negative association between COD and axis 2. COD values should be low for large positive scores on axis 2, and high for large negative scores on axis 2. As a result, it was assumed that Chironomidae would be abundant at low COD sites, while Diptera and Corbicullidae would be abundant at high COD sites. Spongillidae and Thiaridae preferred high NH_4 whereas the Sphaeriidae, Gomphidae, and Lymnacidae were

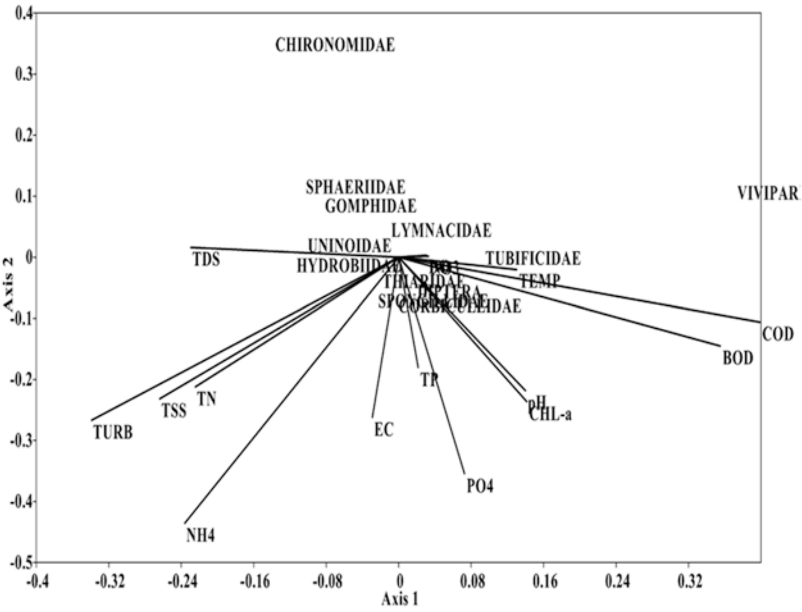

Fig. 7.6 CCA for investigation of the relationship between macroinvertebrate assemblages and physicochemical variables in sampling sites

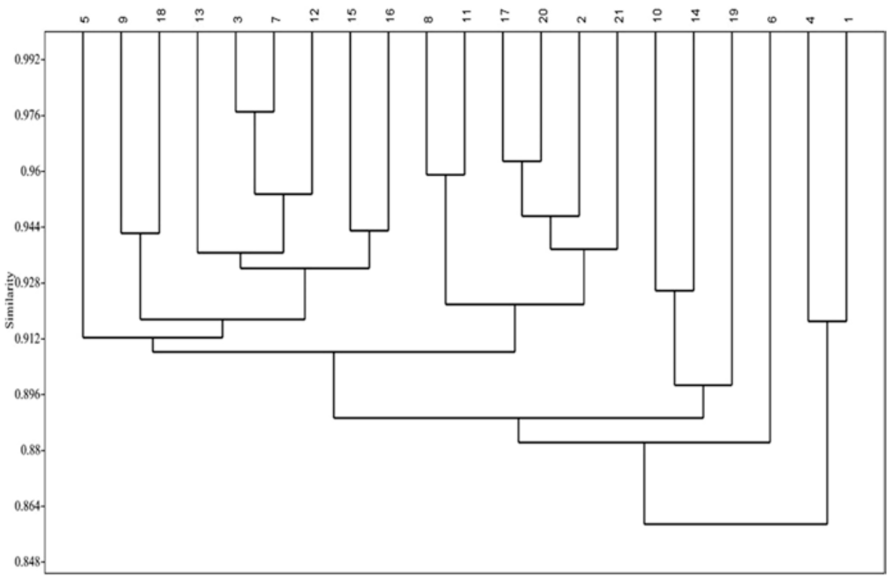

Fig. 7.7 Dendrogram focused on Bray–Curtis' distances from the Mahanadi River's macroinvertebrate population

abundant at low NH_4, turbidity, and COD. Bray–Curtis distance range cluster study of macroinvertebrate populations at the family level yields similar and important associations. A maximum of 97% of association was observed between the downstream regions of Shivnath River (S2) and the upstream region of Jonk River (Fig. 7.7).

3.6 Biological River Health Evaluation Based on Biological Integrity Index (IBI)

The Biological Integrity Index (IBI) is designed to evaluate the health of rivers (Table 7.8). There were three major groups for the eight metrics (M_1–M_8) (species richness and composition, trophic composition, and fish abundances and health condition). The IBI values of node sites (averaged 26) fall under the 3–5 category. According to An et al. (2006), the river's health was classified as good (G). The IBI model values of downstream regions (24–26) were of fair (F) category while that of the upstream region (18) was lower than the other two sites (Table 7.8). Upstream biological river health was poorer than node points and downstream regions. The riffle-benthic species, native species, and deformities among the fishes were the main causes of deterioration. Chemical degradation and nutrient-rich effluents from wastewater treatment plants along the tributaries caused the metrics to be poor.

Table 7.8 Biological Integrity index (IBI) to assess the biological health of Mahanadi River (node point; upstream and downstream to the node)

Category	Metric	Scoring criteria			Mahanadi River watershed		
		5	3	1	Node	Group I	Group II
Species richness and compositions	M_1: Total number of native species	Expectations of M_1 vary within river			13(5)	10.5(3)	11.8(5)
	M_2: Number of riffle-benthic species	Expectations of M_2 vary within river			3 (3)	2 (1)	2.5 (3)
	M_3: Number of sensitive species	Expectations of M_3 vary within river			1 (1)	1 (1)	25 (5)
Trophic compositions	M_4: Proportion of individuals as tolerant species	<5	5–20	>20	88 (1)	91 (1)	50 (1)
	M_5: Proportion of individuals as omnivore species	<20	20–45	>45	13 (5)	25 (3)	50 (1)
Fish abundance and conditions	M_6: Proportion of individuals as insectivore species	>45	45–20	<20	38 (3)	37 (3)	25 (3)
	M_7: Total number of native individuals	Expectations of M_7 vary within river			238 (3)	185 (3)	256 (3)
	M_8: Percent individuals with anomalies	0	0–1	>1	0 (5)	2 (1)	0.6 (3)
Final IBI scores					26 (F)	16 (P)	24 (F)

The health of the river was related to its population structure (fish diversity as well as tolerant and trophic organisms). The Mahanadi River yielded a total of 4308 fishes from 42 different species. *Mastacembelus armatus* (29%) was the dominant species followed by *Xenentodon cancila* (19%) and *Noemacheilus botia* (10%). The dominant fish species were more tolerant/less sensitive species in the physical habitat of the river. The important dominant species at the nodes were *Mastacembelus armatus* and *Noemacheilus botia* (together 66%) and were sensitive and insectivorous (Table 7.9). At node points and upstream regions, the relative abundance of less sensitive or tolerant species was higher than the other two categories (moderate and high sensitive species). However, the number of sensitive species was comparatively higher in the downstream. The presence of insectivorous species was high at confluence points. The highest number of omnivores was present in the downstream region (Fig. 7.8).

Table 7.9 Dominant fish species in the Mahanadi River at the sampling sites River (LS—low sensitive, MS—moderate sensitive, HS—high sensitive, I—insectivores, C—carnivore, O—omnivore)

Location	Dominant species	Sensitive species	Trophic species	Relative abundance (%)
Node sites	*Mastacembelus armatus*	LS	C	47.8
	Noemacheilus botia	LS	I	17.7
	Xenentodon cancila	LS	I	10.5
	Sperata aor	LS	C	5.8
	Sperata seenghala	LS	C	4.6
	Lepidocephalichthys guntea	LS	I	4.4
	Ompok pabda	MS	O	3.7
	Parambassis ranga	LS	C	2.6
Group I	*Mastacembelus armatus*	LS	C	34.36
	Salmostoma bacaila	LS	I	14.28
	Barilius bendelisis	LS	I	12.7
	Anabas testudineus	LS	O	10.5
	Xenentodon cancila	LS	I	9.4
	Channa punctatus	LS	C	7.1
	Channa striatus	LC	C	5.4
	Noemacheilus denisonii	LS	O	1.8
Group II	*Xenentodon cancila*	LS	I	36.69
	Catla catla	HS	O	17.82
	Labeo rohita	HS	O	10.23
	Channa punctatus	LS	C	6
	Ompok bimaculatus	MS	O	5.8
	Mastacembelus armatus	LS	C	5.7
	Noemacheilus botia	LS	I	2.2
	Ompok pabda	MS	O	1.7

4 Conclusion

Given the current situation of increasing pollution that leads to the catastrophic decline of water quality in different water bodies including rivers, our understanding of different water variables and their relationship is essential. River Mahanadi is said to be the lifeline of the states Odisha and Chhattisgarh, therefore basic information is necessary to predict the health of this river. The present study was conducted at the confluences of River Mahanadi with her seven tributaries in order to determine the sources of pollution and their effect on water quality. Water quality analysis via PCA and CA, water quality index (WQI), chemical health via nutrition pollution index (NPI), physical environment health (QHEI), biological health (IBI), and macroinvertebrate diversity were used to evaluate the health of the river. Our results indicated that the river Mahanadi is moderately polluted. The present findings have implications for river quality evaluations and surveillance as they provide the baseline information on various aspects that may help to make better decisions for conservation action plans.

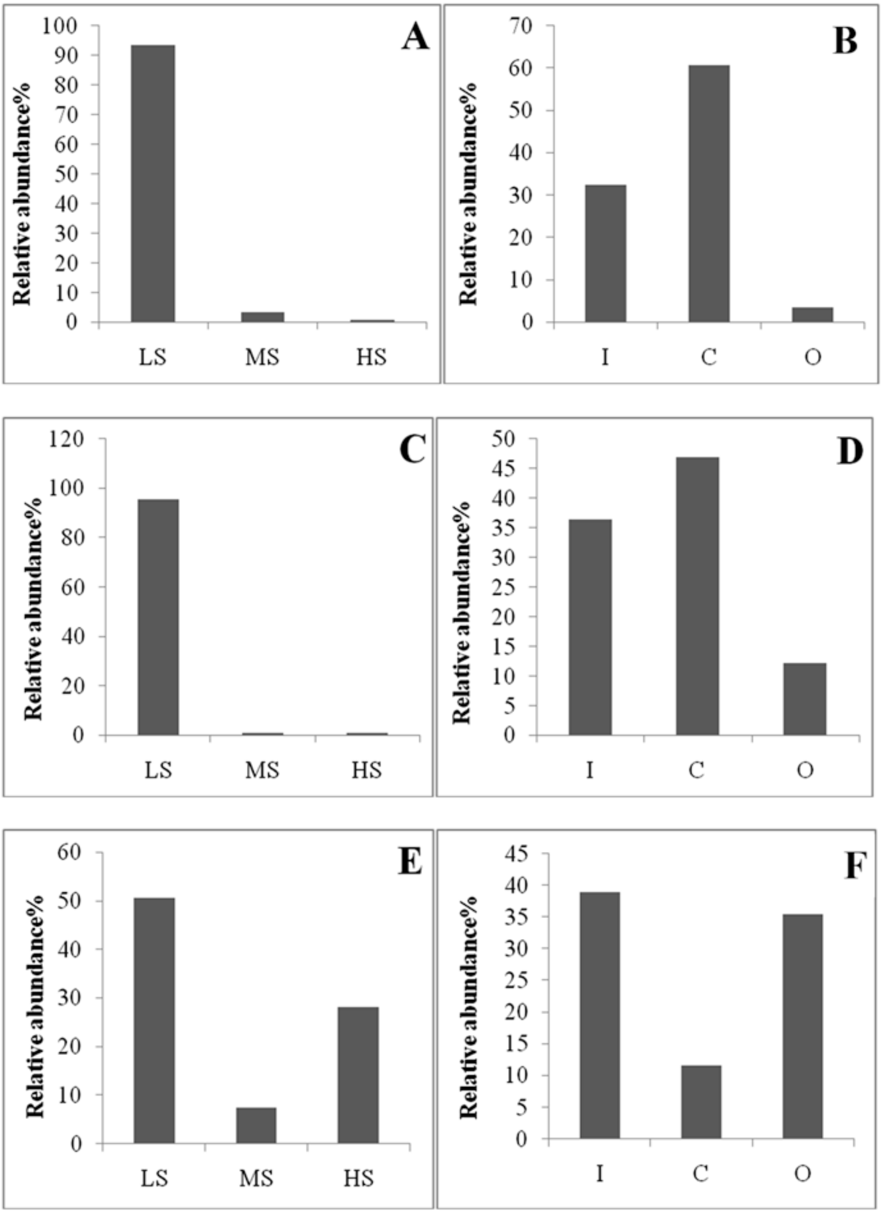

Fig. 7.8 Comparison of sensitive and trophic species analysis in Mahanadi River (*LS* low sensitive, *MS* moderate sensitive, *HS* high sensitive, *I* insectivores, *C* carnivore, *O* omnivore); A and B—node group, C and D—group I and E and F—group II)

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Chapter 8
Estimating Water Quality of Sundarban Coastal Zone Area Using Landsat Series Satellite Data

Ismail Mondal, Sandeep Thakur, Amlan De, Jatisankar Bandyopadhyay, and Tarun Kumar De

Abstract The study is conducted to explore the impact on geomorphological landscapes of the present form to compactness in groundwater, surface water, and salinity. Now the study to validate from the in situ data with the satellite-based application has been used to prepare the surface water quality delineation map. In various landform topographies, features have been recognized on the source of the spectral signature of the Landsat satellite TM and OLI sensor image and to extract in all estimate water bodies, i.e. salinity, NDTI, and NDWI with validated from the in situ water quality analyser instrument and laboratory analysis. The study was assumed to measure the periodic and spatial differences in Chlorophyll, Turbidity, pH, DO and Salinity are compared with the literature values. The present study reveals the variation of the physico-chemical phenomenon of the surface water, this determination to receding tide and flow to transformation salinity and temperature as seasonal fluctuations. The analysis report is highlighted that salinity and turbidity have been shown to a high concentration during this period, overall concentration of the water quality parameters was governed by flushing of the rainfall, river water flow, coastal and inland waterbodies intrusion runoff from the agricultural fields. Turbidity

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content value was higher 100–245 ntu as the Hooghly, Matla, Ichamati, Kalindi, and Bidyadhari of the lower stretch, and salinity value of 25.4–29.7 psu that the parallel variation in a high amount of suspended sediment continuing, are respectively.

Keywords Water quality · Surface water delineation · Coastal process · Landsat TM and OLI

1 Introduction

The Sundarban coastal region biosphere remains below cumulative pressure due to the expansion of industrial production, trade, tourism, and subsequent humanoid population development and immigration, and worsening water quality (Banerjee 2013). This zone is characterized by very high biological and ecological productivity which are constituents of the worldwide life span system. Marine environmental productions (Ascoli 1870–1920) are an energetic part in adaptable microclimate and foremost sink of carbon and oxygen foundation. The growth of industrial coastline has occasioned in the squalor of coastal environments and lessening the alive possessions in the Exclusive Economic Zone (EEZ) has to form in marine and coastal biodiversity and its productivity. Episodic extreme events such as floods and cyclones are the stances of the thoughtful hazard to anthropological activity in human life and belongings to the coastal area (Mukherjee et al. 2008).

The surface water quality estimation is an important task to obtain quantitative and qualitative information on the characteristics of the coastal and inland water bodies. There are multiple challenges for in situ measurements and monitoring of the water quality estimation can be excessively expensive and also time-consuming. Another method is to monitor and estimate the water quality through satellite-based image analysis. In the previous years, remote sensing and GIS techniques and its abilities have been studied for observing in different water quality parameters (Yigit Avdan et al. 2019). At present, the remote sensing is an operational platform for estimation of the various parameters, i.e. surface water level monitoring (Crétaux et al. 2011), synoptic soil moisture content assessment (Su et al. 2003), demand as water quality modelling and monitoring (Ines et al. 2006) groundwater estimation, flood mapping, and management. The present study depending on various satellite sensors has been used for the mapping of the surface water bodies and its monitoring. Therefore, the moderate spatial resolution data from the Sentinel-3, Ocean Color sensors have been used for different water quality applications, i.e. inherent optical particle's Chl-a, TSM, CDOM, etc. (Amitrano et al. 2014; Kaplan and Avdan 2017a, b), and the multispectral sensors of Sentinel-2 have been used for extraction of all inventory water bodies and also monitoring. The MODIS Ocean Color sensors have been used for water quality estimation (Yigit Avdan et al. 2019). Besides, Unmanned Aerial Vehicle (UAV) data were used for water quality measurements (Koparan et al. 2018). Similarly, the Landsat series data were used for measuring detailed water quality parameters, i.e. chlorophyll-a, water transparency, turbidity, and suspended sediments (SPM) (Dogan et al. 2016). The estimated water quality parameters help in decision-making vis-à-vis the further use of the water quality application and modelling.

Finally, the latest technology Remote Sensing and GIS applications are tremendously appreciated to the expansion of databases to examine combined ways and originate management exploit plans. In the present study, we are using the multispectral satellite data of Landsat series that have been applied in different marine and coastal environment assessment. The GIS and Remote Sensing has been used to assimilate in satellite-based info to detect the inland aquaculture locations, coastal instruction, and EEZ of the Sundarban deltaic region. The main objective of this study is to classify, create, and measure all inventory in water quality for water resource data, development, and management. The salinity and turbidity water quality data validated the in situ measurement through satellite-based observations regarding the water quality parameters. The land surface water body was extracted from the TM and OLI time-series images with the help of the Normalized Difference Water Index (NDWI) model. Physio-chemical measurements of the Sundarban estuary system such as pH, DO, Turbidity, Salinity, and Chlorophyll Concentration portray that water quality is depleting due to various anthropogenic activities in the delta. Therefore, the evaluation of the effectiveness of water quality management is essential in the context of the Delta. Thus, the present study is an investigative data analysis with the Water Quality Analyzer Instrument component with reference to seasonal distribution of water quality and computing its trend analysis.

2 Materials and Methods

2.1 Study Area

The Sundarban deltaic region is situated in Ganga-Brahmaputra-Meghna (GBM) delta that portrays the world's largest mangrove dominated forest in the Bay of Bengal. The entire Indian Sundarban measures a coastal length of 130 km and it extends from …. to 21°30 to 22°10N and from …. to 88°10 to 89°10E (Fig. 8.1). The southeastern part of the two districts of West Bengal—North and South 24 Parganas—constitutes the study area (Mondal and Bandyopadhyay 2014a). The Ganga River has bifurcated into two branches—one flowing eastward as the Padma River in Bangladesh and another flowing southward through West Bengal as the Bhagirathi-Hooghly River (Islam and Guchhait 2020). The Sundarban delta embodies the major tidal rivers that are identified as nursing grounds for different types of estuarine and marine activity. The nutrient concentration of this estuary of the northeast Bay of Bengal is as high as the influx of the riverine system and the differences of mangrove litter. The Sundarban mangrove dominated by estuarine riverine systems acts as a source of various water quality parameters and also nutrients. On the

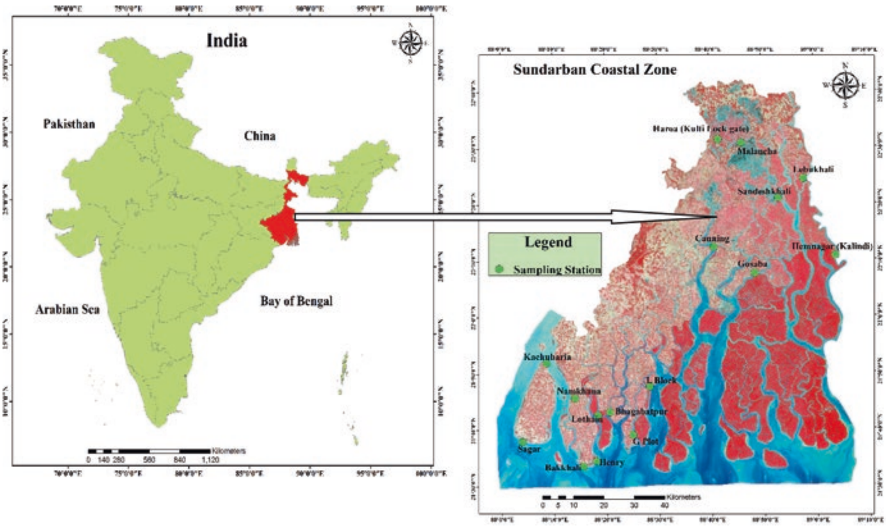

Fig. 8.1 Location map of the study area

Table 8.1 Path and row of satellite images with date of the acquisition

Satellite data	Band	Path/row	Date of acquisition	Resolution (m)
Landsat TM	08	138/45	03/28/1990	30
Landsat OLI	11	138/45	03/28/2018	30

basis of the concentration of litter, Chl-a, and suspended particles, the availability of the nutrients has been measured for the estuarine and riverine network system (Thakur et al. 2019, 2020).

2.2 *Data and Methodology*

The data for the present study are mainly concerned with the satellite data. The satellite-based spectral reflectance range acquired in the different spectral bands (0.40–0.50 μm (blue), 0.50–0.60 μm (green), 0.60–0.70 μm (red), 0.70–0.90 μm (near-infrared), and 9.0–14.05 μm (thermal infrared)) were used to quantify the water quality estimation of interest. Here, we have used the Landsat Thematic Mapper (TM) and Operational Land Imager (OLI) data during 1990–2018 for coastal water quality studies (Table 8.1). Besides, to validate the results derived from the satellite image processing water quality has been measured at 16 sampling stations using the water quality analyser and laboratory (Fig. 8.1; Table 8.2). For carrying out the present work, a systematic methodology has been followed that starts with the collections of satellite data and ends up with the validation of the results (Fig. 8.2).

Table 8.2 Water quality in situ measurement data

Station	Lat	Long	pH	Salinity (psu)	DO (mg/lt)	Turbidity (NTU)	CHL(mg/m^3)
Henry	21°33′57.01″N	88°17′37.17″E	8.2	27.5	6.9	45	6.94
Bakkhali	21°33′31.49″N	88°16′3.98″E	8.2	26.2	5.5	80	7.91
Namkhana	21°45′39.98″N	88°13′53.55″E	8.3	27.3	5.4	150	10.23
Sagar	21°38′33.00″N	88° 3′27.90″E	8.2	27.3	6.41	240	1.1
Kachubaria	21°52′33.01″N	88° 8′6.36″E	8.6	26.8	5.64	220	10.05
Bhagabatpur	21°43′26.17″N	88°18′40.15″E	7.5	28.4	5.3	190	5.16
Lothain	21°41′35.77″N	88°18′37.18″E	8.3	29.7	5.64	130	1.7
G Plot	21°42′8.10″N	88°25′56.56″E	6.9	13.8	4.83	78	1.39
L Block	21°44′21.04″N	88°27′25.29″	7.7	15.7	5.74	140	2.5
Gosaba	22° 8′51.44″N	88°50′49.31″E	7.9	17.7	6.86	100	3.5
Hemnagar (Kalindi)	22°12′24.45″N	88°58′47.28″E	7.4	21.7	5.48	245	4.3
Lebukhali	22°21′58.83″N	88°57′30.65″E	8.2	21	4.9	230	3.7
Sandeshkhali	22°21′2.18″N	88°52′47.25″E	9.3	21.4	5.8	240	5.3
Canning	22°17′13.05″N	88°40′29.34″E	8	12.8	5.6	68	8.9
Haroa (Kulti Lock gate)	22°32′34.73″N	88°40′34.36″E	7.6	7.4	8	100	25
Malancha	22°30′31.54″N	88°46′29.70″E	7.8	15.4	7	130	15

In the present study, satellite images have been used for water quality mapping using various models as follows:

For the purpose of following multi-spectral Landsat OLI data were used for image processing and salinity delineation of the present area. During the pre-monsoon period, salinity has got to accumulate on the land surface due to high rates of evaporation. The water quality reports were collected and utilized during analysis on the basis of in situ measurement. Salinity is a natural characteristic of the soil, but the salinization is caused by the specific activity of the people. Salinity is distinct from the accumulation of the Salt soluble in soil (Dehni and Lounisb 2012; Khan et al. 2001).

$$\text{Salinity} = \text{SQRT}\left(\text{Blue} * \text{Red}\right) \tag{8.1}$$

The Normalized Difference Turbidity Index (NDTI) originally intended to define the turbid water, when it increases the coastal water bodies has been converted to muddy and the spectral reflectance have a comparable to the bare soils, therefore it related to the presence of water bodies (Dambach 2012). The NDTI model has been created in ERDAS Imagine image processing software (Gao 1996).

$$\text{NDTI} = \left(\text{Red} - \text{Green}\right) / \left(\text{Red} + \text{Green}\right), \tag{8.2}$$

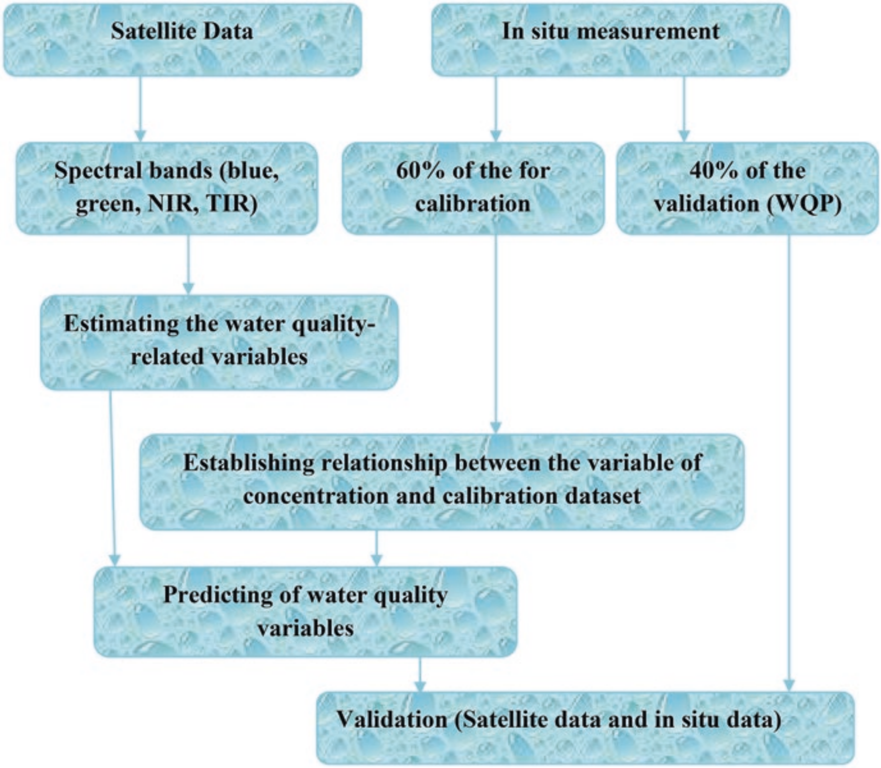

Fig. 8.2 A remote sensing-based framework for the measurement of water quality estimation

The Normalized Difference Water Index (NDWI) model proposed by McFeeters in 1996 detects the landsat surface waters in coastal environments and it is used to measure the surface water volume. The NDWI index has suggested into two near-IR channels; one centred about spectral at 0.86 µm, and the other spectral is 1.24 µm. Following the simplicity NDWI is defined as (McFeeters 1996):

$$\mathrm{NDWI} = (\mathrm{Green} - \mathrm{NIR}) / (\mathrm{Green} + \mathrm{NIR}) \tag{8.3}$$

Whereas the Green Band is the top atmosphere albedo (TOA) of green light has reflectance and the NIR Band is the TOA (NIR) reflectance. McFeeters (1996) emphasized that NDWI values greater than zero are expected to signify the surface water, although the fewer values than, or equivalent, to zero are expected to be the non-water of the surfaces. The NDWI values were calculated from the OLI image using Eq. (8.3); it has been built in Erdas Imagine model maker. The NDWI result was Sediment-laden water, which delineates areas of shallow water in the Sundarban Coastal region.

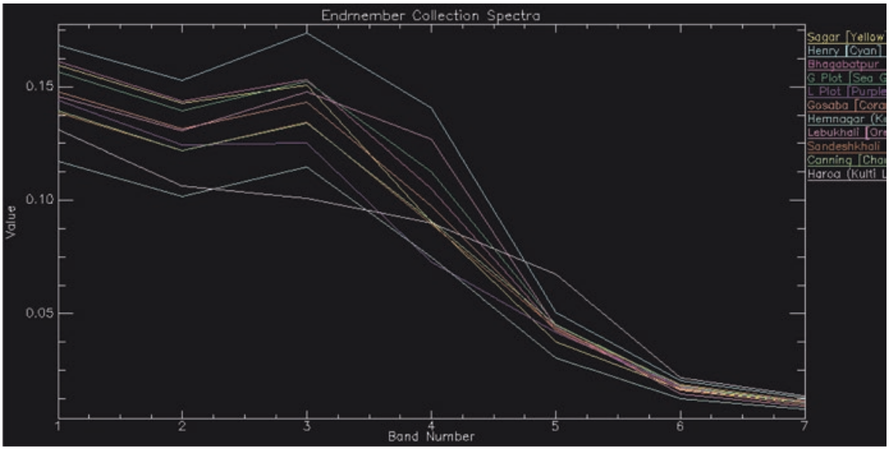

Fig. 8.3 Spectral profile of surface water visible range

In its fluid state, waterbodies has comparatively in a low reflectance, with the clear water having in reflectance is high in the blue spectral range of visible range of the spectrum. The waterbodies have a high rate of absorption and almost no reflectance in the NIR wavelengths region. The turbidity water reflectance is higher in the visible spectrum region of clear water. In fact for the waters containing is higher chlorophyll concentrations. Furthermore, the spectral profiles of the surface water bodies in the Sundarban coastal zone are influenced because most of the atmospheric energy are absorbed. For example, in the visible range, little energy is reflected of the Water quality parameter; Shallow vs. Deepwater; Clear vs. Turbid water and Rough vs. Smooth. The Range of NIR (0.7–0.18 μm) (Fig. 8.3) id totally absorbs and it useful for delineating the surface water bodies, i.e. Chlorophyll, Phytoplankton, Algal bloom, or effects in the reflection.

3 Results and Discussion

3.1 Surface Water Resource

In the present area, the main sources of surface water resources basically include tank and tidal creeks. However, the water of the tidal creeks and tanks is extremely saline, and very hard with the $CaCO_3$ ranging from 2100 to 4900 ppm. During the monsoon period, the salinity level of water decreases due to a reduction in the atmospheric freshwater and, in turn, converted into brackish water which has been observed during the in situ survey with some wrinkled tanks in the Sagar Island. This rainwater has been used for the cultivation of *Rabi* crops in this deltaic region. However, the limited availability of surface water poses a serious threat for various activities including developmental and planning harnessing natural resources. Thus, the crop cultivation in the present area, under the severe stress of freshwater, has to be dependent on rainwater harvesting. To initiate that process, delineating surface water bodies is essential which has been done using the satellite-based model of Normalize Difference of Water Index (NDWI).

Therefore the present study shows that the total volume of surface water bodies was 339,095.5 b.g.l in 1990 (Fig. 8.4) while 360,717.1 b.g.l in 2018 (Fig. 8.5) with an increase of water volume by 21,621.6 b.g.l (Fig. 8.6) over a span of 28 years. This happened because of the conversion of the agricultural land to aquaculture land that

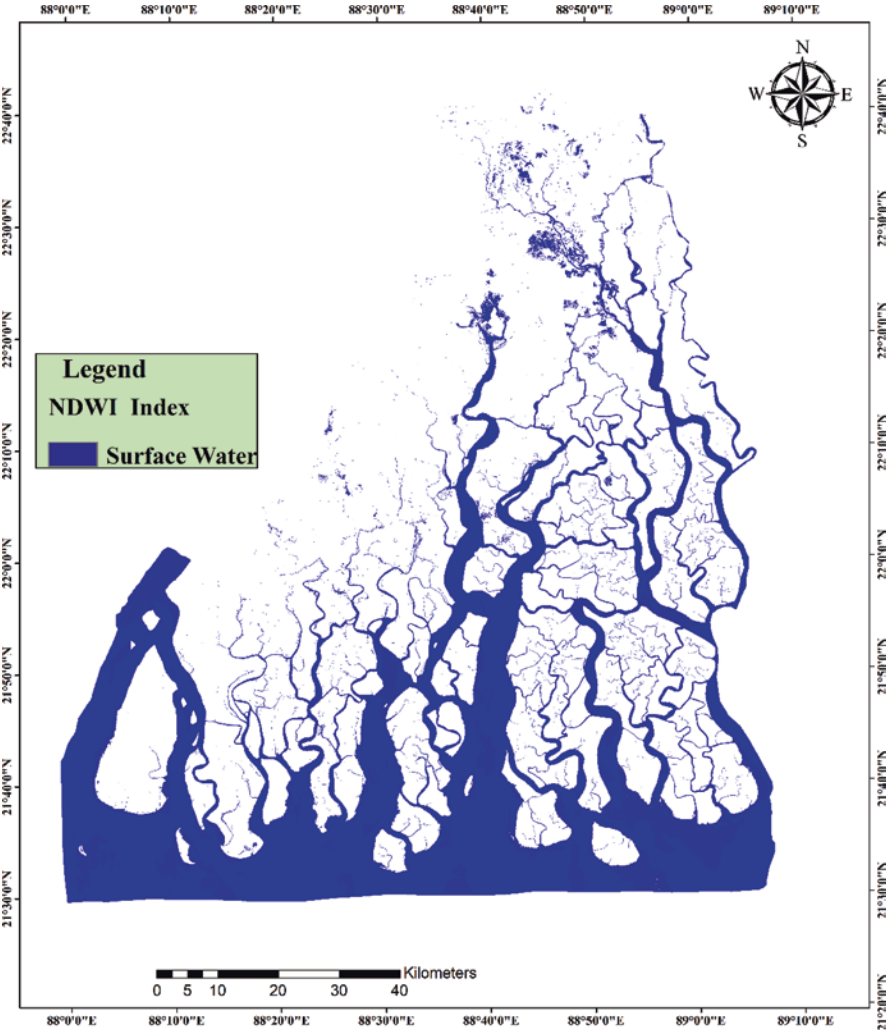

Fig. 8.4 Surface water body of Sundarban (1990)

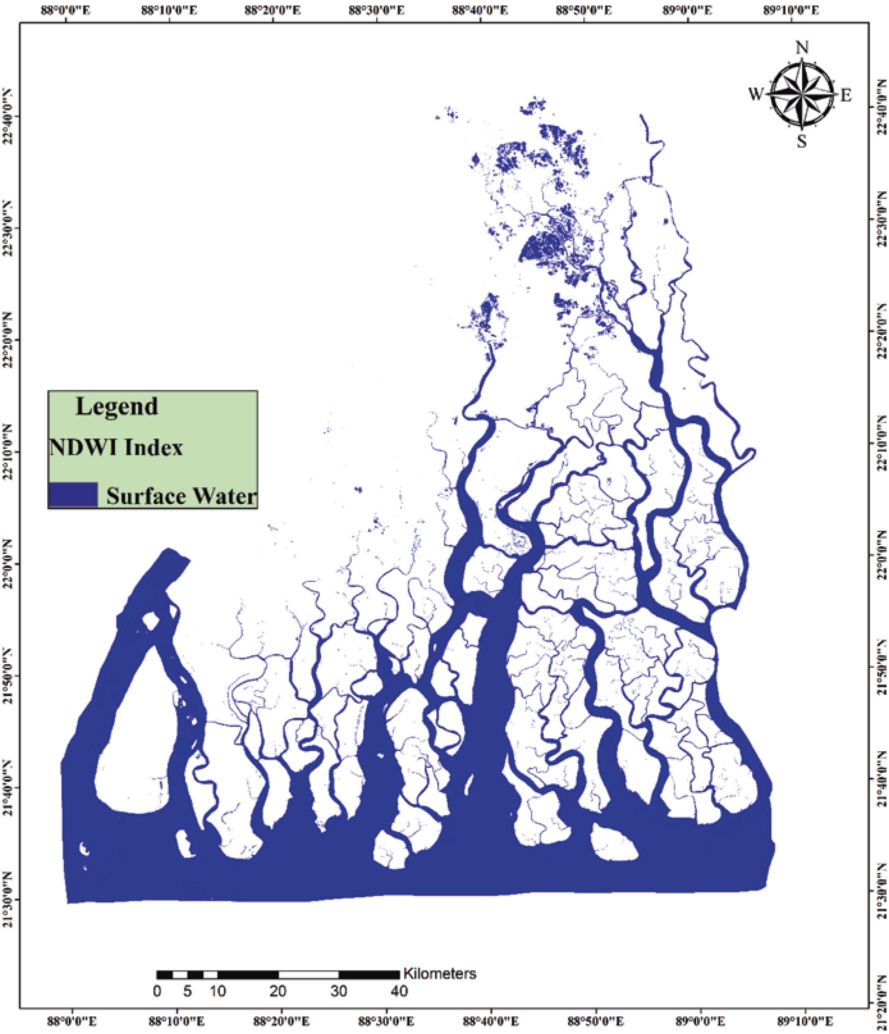

Fig. 8.5 Surface water body of Sundarban (2018)

has been highlighted in maximum part of the Community Development (CD) Blocks like Sagar, Bashirhat-II, Minkhan, Haroa, Sandeshkhali-I and II, and Canning. Therefore, the present study area is characterized by the high fluctuation of seasonal as well as a diurnal flow of water, tidal introgression, and river geometry, which in turn are responsible for repeated changes in the riverine courses of the Sundarban river system to adjust to the morphogenic variables because a slight change in these variables may alter the channel planform and geometry (Islam and Guchhait 2017). This process is active through a continuous erosion and high sedimentation scenario near the Hooghly estuary, Malancha, and Kalindi (Mondal et al. 2016a, 2020).

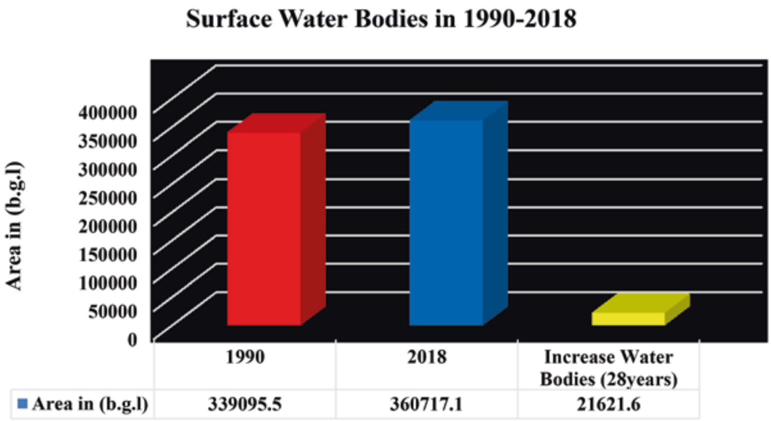

Fig. 8.6 Dynamics of the total area of surface water during 1990–2018

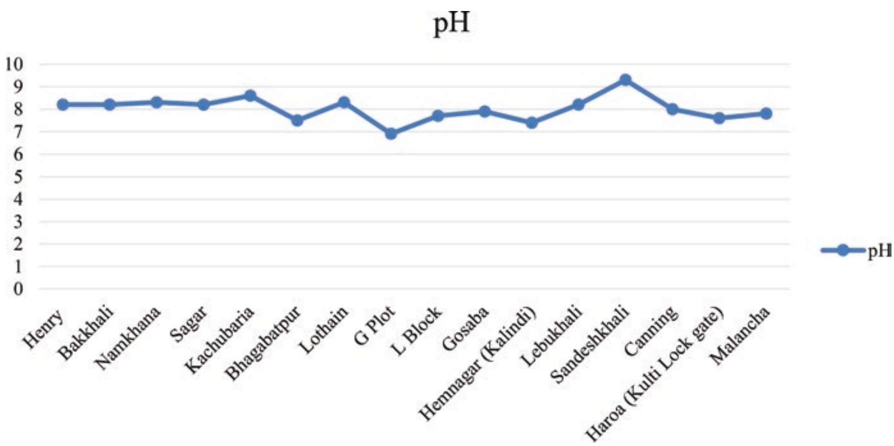

Fig. 8.7 Range of pH

3.2 Physico-Chemical Analysis

3.2.1 pH

The reaction of pH is a major factor to influence the mangrove ecosystem and other biological activities. Here, the pH reaction rate influences the growth of microbial organisms in the mangrove and controls the bio-degradation of disorders and the microbial arbitrated biochemical responses. The microbial degradation, a chemical reaction of biodegradable materials, occurs under the favourable pH range (6.9–8.1). The pH fluctuates between 6.9 and 8.6 that partly obstructs the growth of mangrove dominated vegetation in the Sundarbans. For example, Kachubaria and Sandeshkhali record average level of pH of about 8.6 and 9.3, respectively (Fig. 8.7). Thus, the pH condition in the study area is generally alkaline and may slightly obstruct the microbial-mediated biochemical reaction.

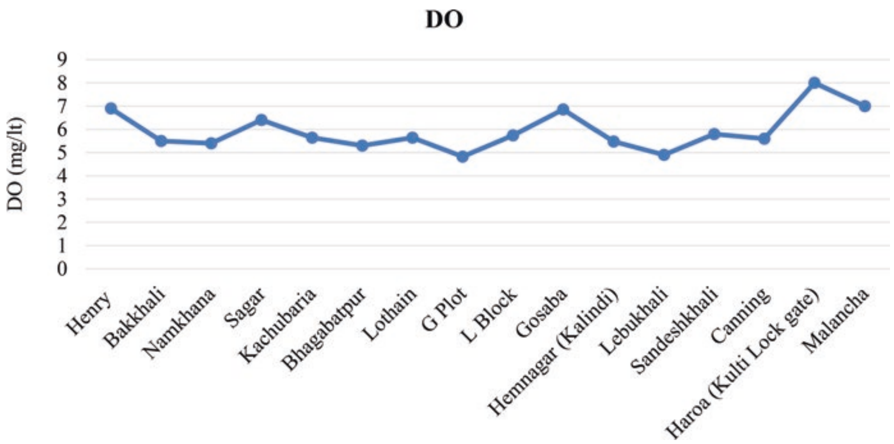

Fig. 8.8 Range of dissolved oxygen

3.2.2 Dissolved Oxygen (DO)

Oxygen is important to all the aquatic organisms, and its accountability for the self-purification methods in surface water bodies varies with salinity, turbulence, temperature, the photosynthetic activity of plants and algae, and atmospheric pressure of the water. The equilibrium solubility of (O_2) in the surface water bodies varies from 2.6 9.8_1 mg/l L1 (8.6 mg/L_1). DO plays an important role in the growth, primary production, and existence of aquatic organisms in the surface water bodies of the Sundarbans deltaic region in different oxidation processes with the physico-chemical interaction. Very high DO of the surface water in some areas (e.g. more than 8 mg/L in Kulti Lock gate) is discouraged spiritually as too much high in DO determination improves the growth of algae, phytoplankton, chlorophyll, and zoo-plankton bacteria foremost to eutrophication which is determined to consume the oxygen layer. In the higher proportion inside the water changing the waterbodies into deceased coastal surface water (Fig. 8.8).

3.2.3 Chlorophyll Concentration

The microscopic plants, called algae, in coastal water, lagoons, and the lake appear green due to the chlorophyll content. The concentration of chlorophyll is high during the summer while low in the winter season because it is not favourable for plants to grow in the winter season. It is noteworthy that numerous human intervention in the form of generation of sewage and obliteration of the coastal wetland and river shorelines affects chlorophyll concentration in coastal water (Horne and Goldman 1994) and it absorption can performance to indicator abundance of phytoplankton and biomass in the Sundarban rivers waters. The natural levels of chlorophyll have been fluctuating in different periods in the study area (Jamshidi and Bin 2011). The measurements show that chlorophyll concentrations have increased with the depth of water in the Sundarban riverine system. The concentration varies from 1 to 49 g/L

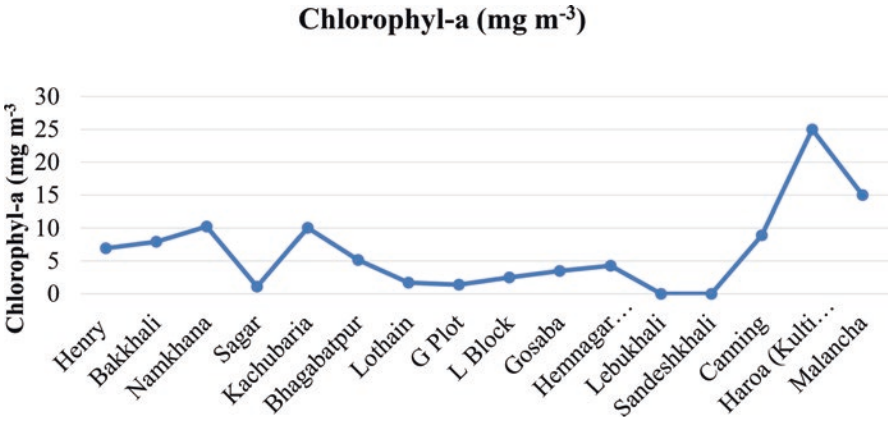

Fig. 8.9 Range of chlorophyll concentration

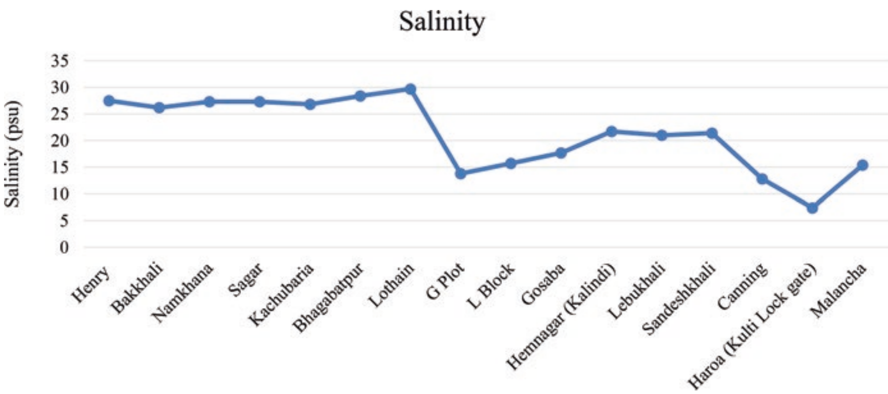

Fig. 8.10 Water salinity range

(Mondal et al. 2016b). We investigated 16 locations to collect the water sample along the Sundarban coastal region. During the pre-monsoon time, maximum concentration value was 25 g/L in Kulti area and the lowest chlorophyll concentration found in Sagar was 1.1 g/L (Fig. 8.9).

3.2.4 Water Salinity Range

The present study executed over the 16 various locations depicts a diverse salinity level ranging from very low (0–7.4 psu) near the Kulti to very high (21–29.7 psu) in Sagar, Bakkhali, Lothian, Kalindi, and Sandeshkhali. However, a moderate degree of salinity (12.8–17.7 psu) is observed at the Malancha bridge, G-Plot, Canning, L-Block, and Gosaba (Figs. 8.10 and 8.11; Table 8.3).

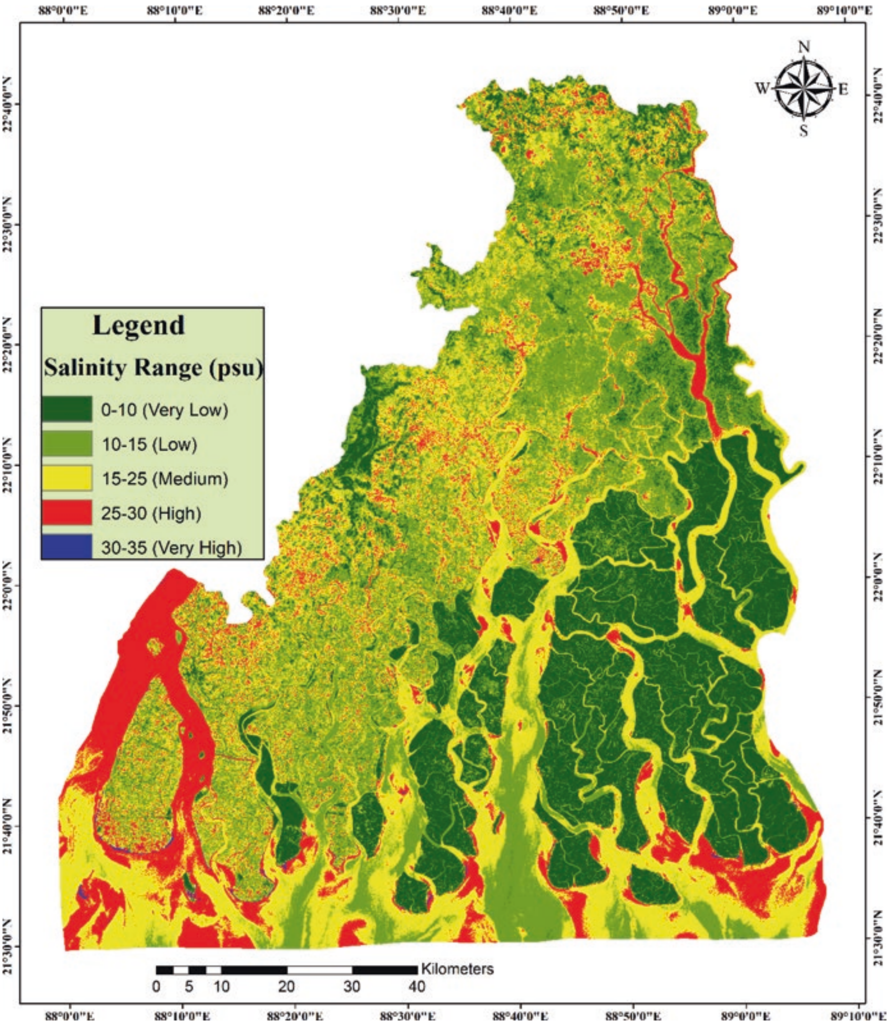

Fig. 8.11 Salinity index map

Table. 8.3 Showing the salinity range value

Salinity range	Water salinity nature	Remarks
7.4–13.8	Represent low salinity	The tidal flat where mud sand both are present for this area
13.8–21	Moderate salinity	Highly mud flat is present and tidal integration
21–29.7	Very high salinity zone	Totally highly tidal fluctuations

As these rivers maintain a two-directional flow pattern, the salinity level also fluctuates depending on the nature of local tidal creek, intensity of river flow pattern, and the severity of extreme weather conditions. In brief, the local regions in the Sundarban delta show an increasing or decreasing pattern of salinity determined by the storm surges and subsequent volume of precipitation.

3.2.5 Turbidity Water

The turbid water map shows that the average ranges of turbidity have been categorized into three zones, i.e. high, moderate, and low, based on the unique pixel value. The high turbidity in the study area was found in Sandeshkhali, Lebukhali, Malancha, Kalindi, Bhagabatpur, and Sagar, which is extended to the Sundarban mangrove evergreen dominated forest. Besides, the moderate turbidity was found in and around the site of Hooghly river mouth. Furthermore, the high suspended sediment loaded (200–250 ntu) condition was noted in the mudflat areas of Henry, G-Plot, Canning, and Hasnabad area, with the value ranging from 50 to 200 ntu (Fig. 8.12). The average turbidity value ranges from 100 to 240 ntu and recorded as high as 240–245 ntu near the Kalindi and Hooghly river mouth area (Fig. 8.13).

3.3 Hydrological Condition

3.3.1 Waterbodies

The surface waterbodies in the Sundarban deltaic system encompass major rivers, tidal inlets, creek, estuaries, and system of complex slightly saline water sequences along with inundated marsh and swamps (SWID 1998). The floods of surface water are commonly found in mangrove dominated vegetation for a long period in the Sundarbans deltaic region. The movement of water bodies with the inundation and ebb tide in the mangrove forest is a steady phenomenon. In the Sundarbans coastal region, river water contains a higher volume of dissolved organic matter and particulate organic matter. Therefore, the water flowing into the mangroves dominated forest becomes rich in nutrients such as of phosphates, nitrates, silicates, and other trace metal components with harmful mixtures like tannic acid and flavonoids giving it a brown colour in oxidized state. The Sundarban Mangrove ecosystem has survived in tidal inundation. Sufficient particulate organic matter is found in the water of mangrove marshlands. The surface water temperature of areas in vicinity of the brackish zone is slightly higher than that of areas connected to the coastal and marine water bodies. Therefore, the mangrove swamplands are diverse through space and time that suggest an exceptional ecosystem of innumerable floral and faunal diversity. The pH, DO, tides, temperature, and salinity are the most important physico-chemical properties of the watercourse of the Sundarbans delta.

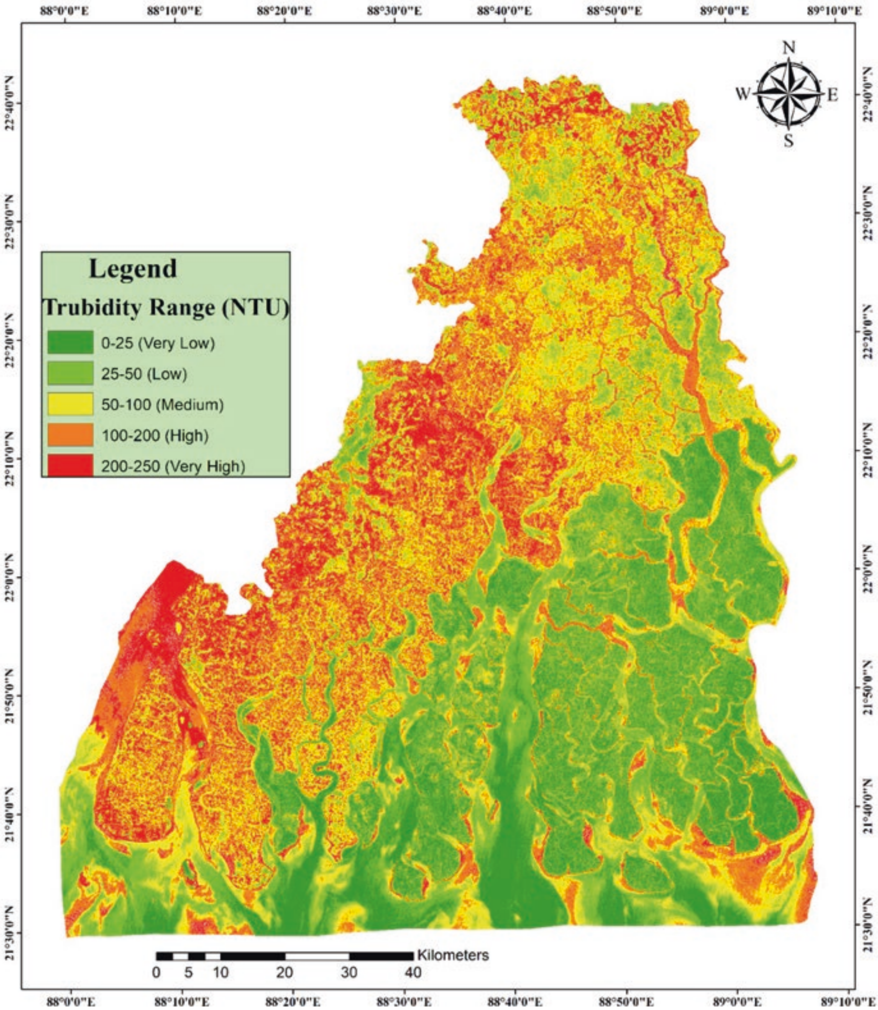

Fig. 8.12 Turbidity index map

3.3.2 Ground Water

The present area is diversified by the presence of fluvio-tidal and the marine coastal facies in sediment deposits (Chakrabarti 1987). Those all deposits were, at large, absorbent in the environment and serve as the source of drinkable groundwater. The lithology of deep tube wells by the survey Central Groundwater Board, West Bengal (CGWB) and the state Public Health Engineering Department (PHED) expose that the aquifers are lenticular and were inter-connected. The granular region records variable depth of about 9–30 m among the depth of 12.5–396 m. Below ground level

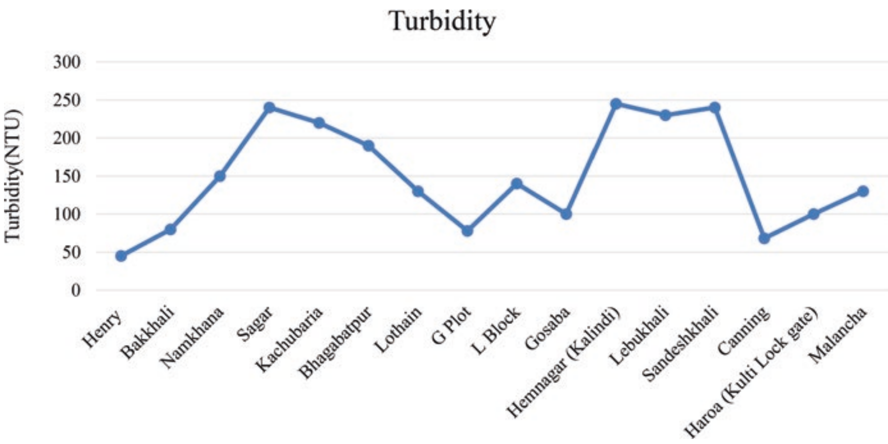

Fig. 8.13 Range of turbidity

(b.g.l), the widespread hydrochemical profile as exposed from the electrical log of the boreholes (CGWB 1988) in Sagar Islands, Kakdwip, and Namkhana portrays a sequence—(1) Saline water—0.00 to 84.5 m b.g.l., (2) Brackish water—84.5 to 161.0 m b.g.l., (3) Fresh water—161.0 to 331.0 m b.g.l., and (4) Salt-water—331.0 to 531.0 m b.g.l.

The deeper aquifers among 180–330 m b.g.l. (CGWB 1988) are the potential source of freshwater influx with the imperative water-saturated granular zone in the depth extending to 250–280 m b.g.l. The potential groundwater in the deeper aquifers transpires under the confined ailment and is lapped by resources of small diameter tube wells fitted with a hand pump for the drinking purpose only (overall depth variable from 202 to 307 m). The chemical analysis shows that samples of water from the deeper aquifers (pH fluctuating in 7.50–8.05 and the total hardness as $CaCO_3$ ranges from 60 to 95 ppm; (Chakrabarti 1992) suggest that the groundwater is as well as freshwater and could be used for the domestic and industrial purposes according to optimum discharge in and around of 2400 m^3/day. It has been suggested discharging 2300 m^3/day as per the tube well. Above the current situation, there looks to be a little opportunity to increase in the cultivated area along with the crop yield as per the unit area. However, it is sensed that this possibility could be increased considerably with the development of the irrigation potentialities through the cumulative use of both surface and groundwater resources because the lesser the irrigation hazards, the higher the agricultural return (Sarkar and Islam 2019). Therefore, to overcome the problems of the salinity, turbidity, and associated issues, some steps may be useful, such as (1) tapping the suitable groundwater resources for local irrigation keeping in mind the limitations of environment and energy. (2) Restoration of the existing drainage pattern and creation of a new drainage pattern, where needed, following the regional slope directing from the north to south (0.20 m in 25 km on average) and the local of micro-level variance (Chakrabarti 1991). This may solve the problems of the prevailing water logging as well as lessening the saline water of the surface water bodies. (3) Rainwater harvesting in small dams, tanks, and run-down into the canals/tidal creeks for irrigation of the *Rabi* crop. Locations for such tanks are to be selected based on the local and micro-relief difference and essential to away from the tidal creeks to avoid the saline water intrusions, mainly during the high tide period. I (4) Construction of channels/dams in a certain appropriate portion of the inactive tidal creeks to facilitate mixing of the ocean water with the rainwater (during in rainy season) which then could be used for the *Rabi* cultivation. Actually, the villagers have already assumed that this system is very local by making some small earthen bunds through the tidal creeks. The freshwater thus obtained is used for *Rabi* cultivation on a small scale. However, its construction of such channels in an active tidal creek should be based on the complete study on the local geomorphology as well as micro-level relief changes, wave/current force of the diurnal tidal flow. Moreover, unscientific construction induces an unbalanced condition in existing natural environments that may lead to severe destruction and abrupt water flushing during spring high/storm high tide in the rainy season (Chakrabarti 1991).

3.3.3 Freshwater Related Problems in Sundarbans

The Sundarbans delta is surrounded by tidal creeks and rivers with ample surface water bodies. The entire Sundarbans deltaic area gets into a tidal subsidy of water two times a day. Despite two factors, there is a huge pressure in the entire Sundarbans concerning accessibility of drinkable water as well as freshwater influx. This is due to the fact that the deltaic rivers are also saline in nature. The groundwater condition of the delta is also saline especially in the superficial aquifers. The saline water is to limit its use for domestic purposes as well as for the agricultural practices. Even though in aquaculture practices has been the use of high salt-water does not continuously crop appropriate outcomes. Here, the problem is arising it further difficult as Arsenic related issues were found to build a crop up into several blocks of the Sundarbans coastal regions. The local people are dependent on groundwater for drinkable as well as domestic purpose use. However, to save the local community, the fresh and drinking purpose groundwater is available in a deeper aquifer layer. In different places, deeper aquifer has been located at a depth level near about 250–331.0-m b.g.l. By the way, for a local man, making arrangements for the extraction of groundwater from such depth of the tube well is not an easy job.

Interestingly, the average rainfall over the Sundarban coastal region is more than 1500 mm. But without a proper structure, planning, and management, the whole rainfall, that is again seasonal by the high concentration only on a few days, becomes wasted by flowing into the adjacent Bay of Bengal. Absence of discharge upland freshwater into these Sundarbans rivers allows the brackish coastal water to move towards the inland. The agricultural field makes agriculture difficult due to the intrusion of saline water. However, aquaculture land in the Sundarban coastal zone requires water with a limited salinity content. Laterally increasing the salinization, the river depths are also decreasing progressively. The navigation system faces a lot of problems, mainly during low tide periods in various areas. Due to the effect on embankment failure and erosion, people lost their land-living properties and are becoming highly vulnerable in the environment of climate change.

In a nutshell, the Sundarban experiences some problems related to issues to understand the dimension of the unavailability of freshwater such as:

1. Limited to upstream freshwater discharge and issue of salinity
2. Contamination of Arsenic in Groundwater
3. Agriculture and Irrigation Problems
4. Fishing and Aquaculture Problems
5. Siltation and Navigation Problems
6. Due to effect on Land Loss and Embankment Failure

3.3.4 Remediation of Water Issues

The Sundarbans people are compelled to depend on the groundwater potential to meet the drinking water demand and also rainwater for the agriculture and various field requirements. Conjunctive use of all those water sources may help for sustainable development management in this coastal region. About these methods may be used to ensure in the freshwater influx in the Sundarbans is described below.

The Sundarban delta has received sufficient rainwater during the monsoon period. Thus rainwater could be preserved to meet the water demand of the Sundarban people. The rainwater harvesting could be utilized for domestic purpose as well as the drinking purpose after its proper treatment (Gayen and Zaman 2013; Mondal 2014; Haza et al. 2015). The major solution to mitigate in the water-related issues of the Sundarban coastal region is a requirement as large-scale rainwater harvesting. Renovation of the existing river, tidal creek, and canals is also essential. It will help to preserve a good capacity of the existing rainwater, and it would be useful for the cultivation purposes. The rainwater harvesting methods also limit the over-exploitation of the groundwater recharge.

Now, the artificial groundwater recharge and also de-salination for the excess of rainwater could be rejuvenated in artificially within the shallow aquifer to decrease of salinity. During that time this method will advance to decrease of salinity into the shallow aquifer it could consequently be utilized for domestic and agriculture purposes (Gayen 2009; Haza et al. 2015). The present study of the deltaic rivers and their siltation and discharge rate is too high (87.1 m^3/s) (Mondal et al. 2019) and also high salinity (>17 psu). We recommend that type of desalination planning method could be tried using silt free less to saline water (5–10 psu) of the shallow aquifer. The groundwater and surface water resource conjunctive use and it will be helpful for the sustainable water resource implication and management of Sundarbans.

Further study can be taken up on the rejuvenation of the Jamuna River with necessary clearance at the offtake point from the Hooghly River (i.e. Tribeni area) and subsequently if the river is artificially straightened and joined with the north-south aligned course of the moribund Padma river (a distributary of the Jamuna river) around to provide upland flow to the Bidyadhari river/Kulti Gang. The next study may be carried out on rejuvenation of the inland waterways transportation through the Bhagirathi-Hooghly River; canal network system of Kolkata may be considered only with the increase of reversible upland flow of Bhagirathi-Hooghly river system through Farakka Barrage. In this regard, time-sequential data may be used for monitoring of the thalweg zone of the trunk river and to identify the blotted out portions of the canals and spill channels of the trunk river in different seasons with future perspectives. Finally, from Farakka Barrage about 40,000 cusecs ft/s of water is discharged into the Bhagirathi system every year in the dry season. However, the Ichamati river system needs at least some amount of water from Bhagirathi River (from Tribeni via… Jamuna to Ichamati and from Jamuna to Bidyadhari system) to save the Sundarbans ecosystem and to restore the valley system of Sundarban rivers.

4 Conclusions

These are the important bounties in nature subsidizing to the nutriment of life on the earth. Sundarbans is the largest mangrove forest in the world and is contributing to the nutriment as well as to the safety of living existence in the delta. Unscrupulous human interference is a major threat to the ecosystem. The surface water mixed with and waste materials which is coming mainly from the Kolkata metropolitan through Bhangar Khal affecting Bidyadhari River has been escalation the salinity and intrusion towards the anthropogenic pressure due to sea-level rise is one major issue to pressures the Sundari, Garan, Gewa Keora and other Heritiera trees, which are now under danger due to increasing the salinity level is very high. The mangrove plants can grow on the soil of the salinity ranges between 3 and 29.7 ppt in the ocean water. The harmful effects of Indian Sundarbans exacerbate throughout the dry periods, the Raimangal river flow is a feeder to supply the freshwater bodies in the Sundarbans, falls severely (Mondal and Bandyopadhyay 2014b). The time-series satellite data were used and to analysis in 28 years span period (1990–2018) and also laboratory exploration of those data is distinct that of the surface water quality variation is abundant in Sundarban coastal region approximately the fluctuations are. Here, diverse landform features were identified based on a spectral signature of Landsat TM and OLI image and it extracts all inventory water bodies, i.e. salinity, NDTI, NDWI, and field investigation the water quality analyser instrument. The water quality of turbidity, vegetation exposure to classify the pond can help as a supervisory tool, in the preservation ordering of wetland of Sundarban (Mondal and Bandyopadhyay 2014c). Finally, we observed that the study area river water salinity has been increased towards upstream to downstream progressively and also increased water and air temperature due to enhanced the land erosion, sea-level rise to intensification in occurrence and concentration of dangerous weather actions, i.e. cyclone and storm, etc. We suggest to the study, to implement into the Integrated Coastal Management Plan or Protected Area Management Plan implement of the study area.

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Chapter 9
Anthropogenic Impacts on Hydro-Morphodynamic Behavior in the Middle–Lower Course of Subarnarekha River, India

Subrata Jana and Ashis Kumar Paul

Abstract The hydrodynamic and morphodynamic behavior in the middle–lower courses of the Subarnarekha hydro system is under threat in the era of Anthropocene. Considering the satellite images, Google Earth images, and field survey records, the present study has been done to understand the anthropogenic impacts on hydro-morphodynamic behavior of the river course and to find out the possible ways of management. The Subarnarekha river in the study area is well recognized for its sand quality. Sand and boulder are intensively excavated from the riverbed position of Gopiballavpur, Rohini, Bhasraghat, Dantan, Sonakania, Jaleswar, and Baliapal sections. Plenty of erosion prevention structures have been constructed at the erosive bank position at different places. The seasonal fair-weather bridges have also been constructed after diverting the natural active flow of the river. All these unscientific anthropogenic activities directly or indirectly create a harsh impact on the natural behavior of the river as well as on the riparian dwellers. The channel margin land erosion and bank failure are accelerated at the position of mining sites and in the downstream section and opposite bank of the embankment structures due to change in hydrodynamic behavior of the river. Despite the worst impact on hydro system, the anthropogenic activities like mining and structural measures need to allow at a certain level after proper study of the hydro-morphodynamic behavior of the river course for the socioeconomic development of the riparian societies.

Keywords Unscientific sand mining · Erosive riverbank · Erosion prevention structures · Hydro-morphodynamic adjustment · Socioeconomic development · Subarnarekha hydro system

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1 Introduction

In the era of the Anthropocene, human existence on the globe is now a challenging issue. The socioeconomic developments of human beings are very much dependent on the nature of the floodplain and fluvial hydro systems in a river valley region (Norris and Thoms 1999). Likewise, in all river valleys in the world, the Subarnarekha river course, particularly the middle and lower courses are significantly affected by the human encroachment within its different reaches. People are enhancing the resource exploration from the river valleys and even encroached into the river course for more exaggerated development. The natural hydrodynamic behavior of the river course is regulated by the seasonal fluctuation of flow regimes (Jana and Paul 2014; Samanta et al. 2018; Jana 2019; Jana and Paul 2020), which is also distorted with the anthropogenic activities. The riverbanks and bank margin areas are regularly affected by the hydrodynamic actions during floods (in the middle and middle–lower course) and diurnal tides (in the deltaic course) (Jana 2019). Riverbank erosion, overbank flooding, and sediment deposition in the floodplain areas are the major problem to the riparian inhabitants. Loss of arable land and other properties, crop damage, and suffering of inhabitants are associated with the flooding. Therefore, people construct embankments and other flood and erosion prevention structures to overcome those problems (Jana 2019). As a result, the natural hydro system is disrupted and crating negative impacts in and around the bank protective structures for maintaining the hydro-geomorphological adjustment.

The natural hydro-morphodynamic behavior of the river stretch has been affected by intensive anthropogenic activities like sand mining from the riverbed and bank margin areas, erosion and flood protection embankment structures, bridge construction across the river, and agricultural activities over the channel bed (Jana 2019). All these are directly or indirectly creating a harsh impact on the hydro-morphodynamic behavior, which is also enhancing the riverbank erosion in the different channel reaches. The intensive and unscientific sand mining activities lead to riverbed drowning and riverbank erosion (Arun et al. 2006; Jana 2019). The hydraulic action is changed due to the installation of erosion protection measures, which can control the riverbank erosion at that particular position, but it intensifies the riverbank erosion in the downstream section and the opposite bank (Jana and Paul 2014). The natural flow direction and hydraulic actions are also diverted due to the construction of cross-channel concreted bridges and other fair-weather bridges, which leads to hydrogeomorphological changes in the riverbed (Allison et al. 2013). The agricultural activities in the riverbed and over the mid-channel bars can create a harsh impact in terms of geomorphological changes and the ecological perspective of the hydro systems (Downs and Gregory 2004). All these activities have been observed in the different channel reaches in the middle–lower and deltaic courses of the Subarnarekha River.

Lots of studies had done in the present river stretch related to the hydrogeomorphological, sedimentological, and riverbank erosion perspectives (Dandapat and Panda 2013; Jana and Paul 2014, 2018, 2019, 2020; Samanta et al. 2016; Jana 2019;

Ilahi and Dutta 2016; Guha and Patel 2017; Samanta et al. 2018), but, the study of anthropogenic impact on hydro-morphodynamic behavior of the Subarnarekha river has been neglected in the earlier works. Therefore, the present study emphasizes the role of different anthropogenic activities in the hydrogeomorphological changes in the different channel reaches of the middle–lower and deltaic courses of the Subarnarekha river.

2 Material and Methods

2.1 Study Area

The study has been carried out between the Jamsola (upstream) and Chaumukh (confluence in the Bay of Bengal) section along the 155 km long stretch in the middle-lower and deltaic courses of the Subarnarekha river (Fig. 9.1). Only the recent river course and its bank margin areas have been considered for the assessment of anthropogenic impacts on the fluvial hydro system. The selected river courses are dominated by the fine sand, silt, and clay materials of Panskura formation and dark gray to black clay associated with mudflats of Basudebpur formation, which were deposited during the Middle Holocene—Late Holocene period (Jana, 2019). The upper course (Jamsola to Baliapal) is dominated by the unidirectional flow with seasonal fluctuation of regimes. Whereas, the downstream section of Baliapal is influenced by the diurnal bidirectional tidal flow with a mean annual range of about 2.31 m (Jana and Paul 2020). The course—medium sand is predominately deposited in the riverbed even up to the Baliapal section concerning the flood dominancy, which promotes the sand mining sites in the different channel reaches. The bank margin floodplain areas have been frequently inundated with the high magnitude flood regimes almost every year in the recent past. Therefore, the fertile floodplain on both sides of the river excessively utilizes for agricultural activities. However, crop damages and degradation of agricultural land due to bank erosion and sand sheet deposition are common events in the study area. Large numbers of people living in the channel margin natural levee positions are affected by the floodwater inundation and bank erosion. In this concern, different types of flood control measures have been taken in the distinctive channel reaches.

2.2 Database and Data Processing

The study has been done based on the different Landsat satellite images coupled with Google Earth images and field-based analysis. The four different multitemporal imageries (path/row: 149/45) of Landsat 5 (TM) of November 14th, 2005 and January 21st, 2009, and Landsat 8 (OLI) of March 28th, 2014 and February

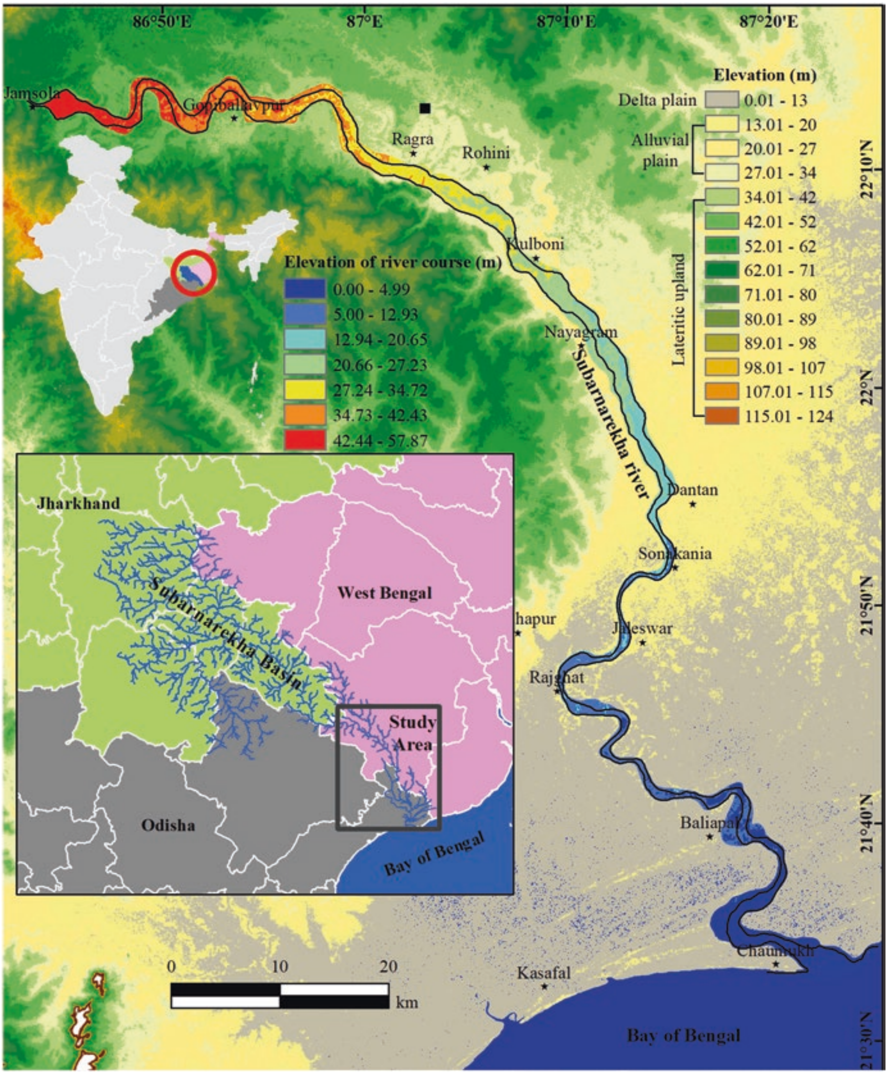

Fig. 9.1 Regional settings of the middle–lower course and delta plain of the Subarnarekha River

27th, 2015 have been collected from the United States Geological Survey (USGS) website (https://earthexplorer.usgs.gov/). Moreover, the GE images of 2007 and 2015 have been used to demarcate and estimate the detailed account of the various structural measures in this study area. The Landsat images have been projected and resampled in the Universal Transverse Mercator projection considering the 45 north zone and world geodetic survey 1984 (WGS84) datum. The images have been co-registered (Beuchle et al. 2015) with <0.5 pixel accuracy of root mean square error (RMSE). The normalized difference vegetation index (NDVI) and normalized difference water index (NDWI) have been used (Ahmed and Akter 2017) to stress out the appropriate spectral response of various surface cover features like active river flow, sand bodies, and bank lines of the river course. The bank lines, sand mining areas, active flow paths, and different artificial structures have been digitized in the ArcGIS 10.1 software. The Google Earth elevation data have been extracted and processed into the ArcGIS 10.1 software to understand the elevation variation of the river course (Nishio and Mori 2015; Maythm 2020). A detailed account of sand mining areas has been collected from the local owner of the mining areas, track driver, and mining labor during the field survey.

3 Results and Discussion

3.1 Impacts of Sand Mining and Artificial Structural Measures

The Subarnarekha river course in the selected study area is renowned for its qualitative supply of sand from the different mining areas of Gopiballavpur, Rohini, Kulboni, Bhasraghat, Dantan, Sonakania, Jaleswar, and Baliapal (Figs. 9.2, 9.3; Table 9.1). Various machines operated intensively coupled with manual labor for sand mining from the river bed (Table 9.1). The sand mining process remains active during October–June at the lean phase of the monsoon depending on the active river flow (Jana 2019). Moreover, different types of erosion controlling embankments and other flood protection structures have been found in the other different bank margin sites, which were constructed to divert the flow direction (Fig. 9.2). The intensive and unscientific mining practice from the riverbed and bank margin areas and embankments structures creates a harsh impact on the natural hydro-morphological and ecological behavior of the river course.

3.1.1 Changing Hydrodynamic Behavior

The excessive sand mining from the riverbed and adjacent bank areas creating a ponding situation that leads to a change in the hydrodynamic behavior of the channel active flow (Fig. 9.4). The hydraulic head, scouring, and turbulent flow pattern are more effective in the depression areas of the riverbed insists erosion of materials from the riverbed and its marginal bank areas (Fig. 9.4). The bank erosion is more intense at the mining sites. The embankment structures have been constructed in the different bank margin erosion-prone areas and its upstream sites to divert the channel active flow (Fig. 9.2). During flood events, the outer bank of the meandering stretch has been heavily affected and eroded by the high flow regime and hydraulic energy. Therefore, embankment protection measures have been taken mostly in the outer bank position of the meandering course (Fig. 9.2) to control erosion and bank

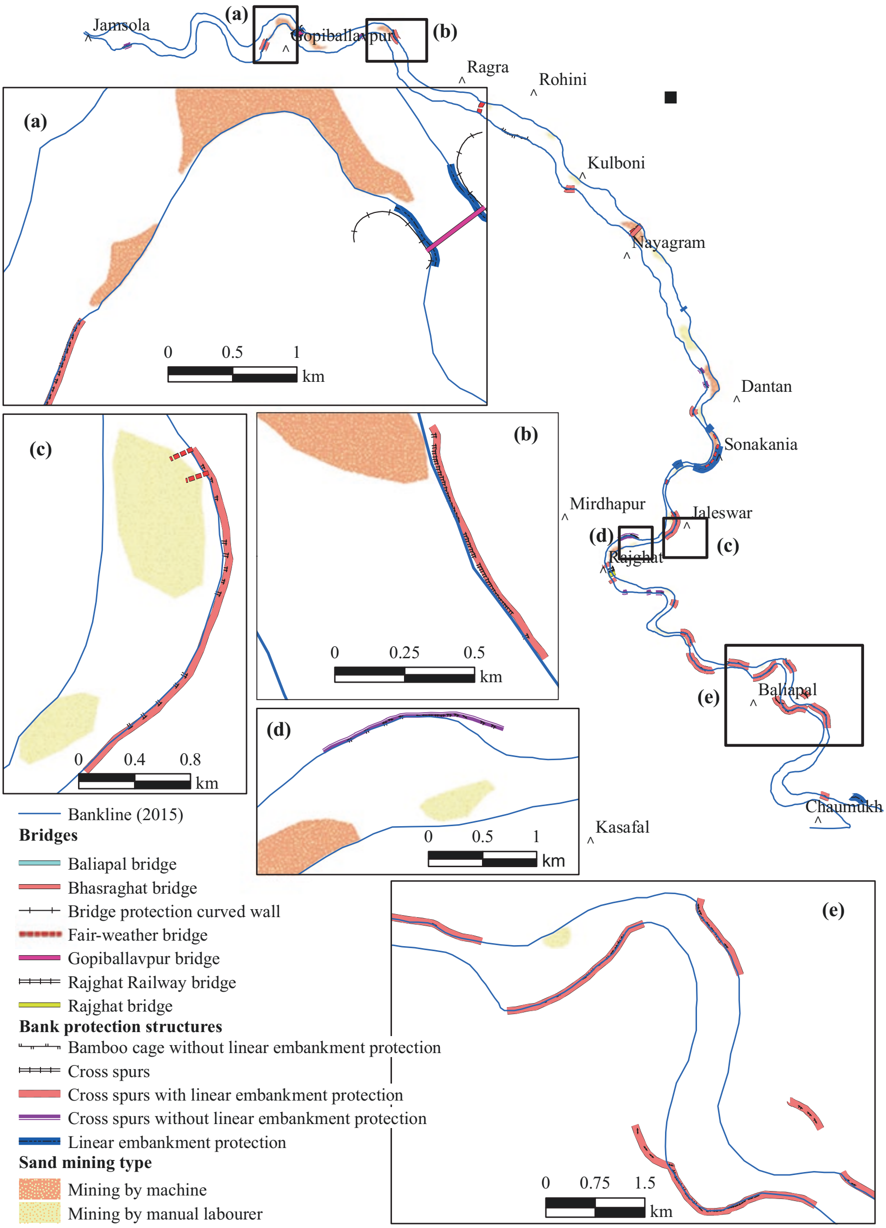

Fig. 9.2 Erosion proactive structures and anthropogenic activities at the different sections of the river course (**a**) Gopiballavpur, (**b**) Mahapal, (**c**) Jaleswar, (**d**) Rajghat, and (**e**) Baliapal sections

Fig. 9.3 Intensive sand mining activities in the different places from the riverbed: (**a**) erosive point bar section near Nayabasan, the boulder quarrying from the riverbed; (**b**) erosive channel margin in the left bank at downstream of Gopiballavpur bridge, the mud-ball formation initiated after bank erosion; (**c**) over the riverbed at Bhasraghat; and (**d**) point bar margin riverbed near Sonakania

Table 9.1 Site-specific sand mining areas and mining processes operated in the river bed

River stretch	No. of the mining station	Sand mining area (km^2)	Mining process
Gopiballavpur	4	2.60	Mining by machine
Rohini–Kulboni	5	1.45	Mining by manual labour
Bhasraghat–Dantan	4	6.23	Mining by manual labour and machine
Sonakania–Jaleswar	7	4.05	Mining by manual labour and machine
Santoshpur–Baliapal	6	1.02	Mining by manual labour

Source: Field survey records from December 2011 to March 2014

failure. In the study section of the Gopiballavpur (near the Dharmapur village), the cross spurs with linear embankment protection structure have been found along the 858 m bank line in the right side of the channel and 14 cross spurs have been placed perpendicularly with the embankment (Figs. 9.4 and 9.5a; Table 9.2). During 2009, the channel active flow has stroked at the position of the linear embankment protection structure (Fig. 9.4). But, channel active flow strikes at the end portion

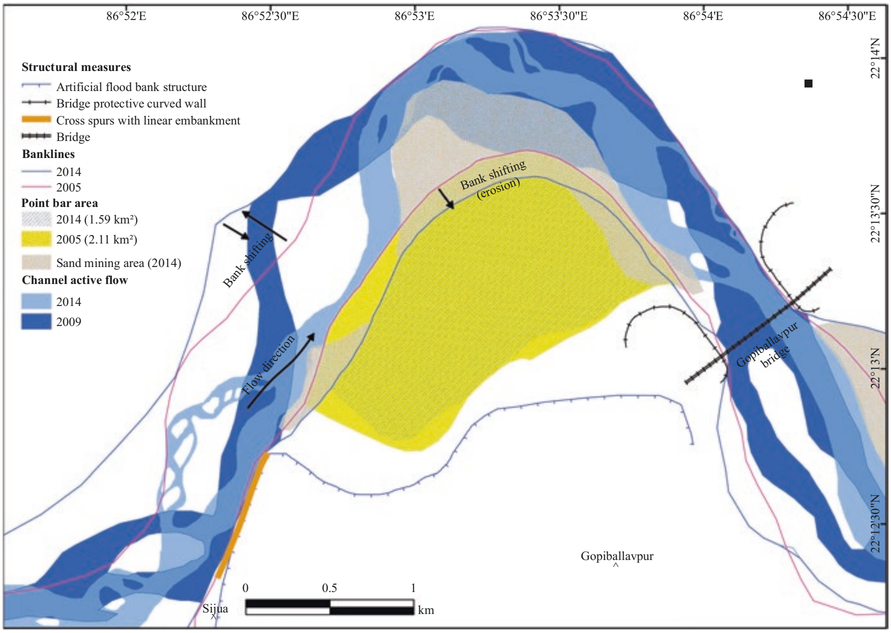

Fig. 9.4 Severe bank erosion at the position of Nayabasan point bar due to intensive sand mining from the riverbed and changing active river flow pattern guided by the embankment structure in the Gopiballavpur section

(downstream side) of the embankment and toward the mining site after the renovation of the embankment protection structure with the addition of cross spurs in 2014 (Fig. 9.4). A similar type of downward shifting of active flow and the hydraulic head has been observed with the synchronized positional shifting of the bank protection wall near the Narashighapur village in the Bhasraghat section (Fig. 9.6). In this site, 12 cross spurs with linear embankment protection have been constructed (in 2015) along the 820 m stretches on the right bank margin (Table 9.2). In 2007, the position of the embankment protection structures has remained in the upstream section (yellow color) compared to the embankment protection structures of 2015 (red color) (Fig. 9.6). During 2007, the river flow has been concentrated at the position of the bank protection wall (yellow color), which gradually shifted toward the downstream section. During 2015, a new bank protection structure has been constructed (red color) in the downstream section of the previous one (yellow color) (Fig. 9.6). The recent image (2016) shows that the flow concentration has been shifted in the successive downstream section of the same (right) bank (Fig. 9.6). However, sediment has been deposited in the subsequent positions of the bank protection structure of 2007 and 2015. Moreover, the concreted bridges and fair-weather bridges have been constructed across the river course at different places (Fig. 9.2, 9.5c-d; Table 9.2) for communication purpose. The fair-weather bridges have generally been constructed with the diversion of active channel flow (by sand and boulder filling in the

Fig. 9.5 Impacts of structural measures on river course. (**a**) Flow diversion and sediment deposition by the cross spurs with linear embankment structures near Dharmapur of Gopiballavpur section, which leads to bank failure in the opposite bank, (**b**) artificial backwater formed due to construction of bridge protection curve wall in the left bank near Gopiballavpur bridge, and (**c**, **d**) diversification of river natural flow for the construction of the fair-weather bridge (before the construction of new “Jangalkanya Setu” at Bhasraghat

riverbed) through a concentrated part of the river course (Fig. 9.5c–d). Both the concreted bridges and fair-weather bridges divert the natural flow and hydraulic actions of the River.

3.1.2 Morphodynamic Adjustment

The intensive and unscientific ways of sand mining and construction of erosion protection structures and bridges are regulating the hydrodynamic behavior of the river, and the sediment erosion and deposition nature are also synchronized with this. During flood events, huge volume of sediments is deposited in the riverbed that is excavated for the rest of the period of a year. However, the channel margin land is eroded during high magnitude flood events. Therefore, the seasonal and long-term erosional and depositional natures of different riverbed morphological features have been found in the studied river section.

The point bar erosion has been observed in the mining site of the Gopiballavpur section (Fig. 9.4). The bar area is gradually shrinking coupled with channel margin land erosion due to intensive sand mining. The bar area was 2.11 km^2 in 2005, which is remained only 1.59 km^2 in 2014 with the reduction of 0.52 km^2 channel

Table 9.2 Details of various types of erosion prevention measures and bridges at the different positions in the middle–lower course of the Subarnarekha River

Place	Types of artificial structures	Length (m)	Bank position	Remarks
Daha Munda	Cross spurs without linear embankment protection	587	Right	7 no. of cross spurs
Dharmapur	Cross spurs with linear embankment protection	858	Right	14 no. of cross spurs
Gopiballavpur bridge	Linear embankment protection on both banks	590	Right	Only bouldering
		548	Left	
	Bridge protection curved wall	1022	Right	
		936	Left	
	Bridge	563	Across the river	Roadway
Alampur Pirasimul	Cross spurs without linear embankment protection	324	Right	4 no. of cross spurs
Mahapal	Cross spurs with linear embankment protection	940	Left	33 no. of cross spurs
Bidyadharpur	Fair-weather bridge	1113	Across the river	Used for public transport
Ramchandrapur	Bamboo cage without linear embankment protection	2444	Right	77 no. of cross bamboo cages
Narashighapur	Cross spurs with linear embankment protection	820	Right	12 no. of cross spurs
Bhasraghat	Bridge	1376	Across the river	Roadway
	Fair-weather bridge	1422		Mainly used for public transport
Kotpada	Linear embankment protection	277	Left	Bouldering along the bank
Makidia	Cross spurs without linear embankment protection	477	Right	4 no. of cross spurs
Mankiria	Cross spurs without linear embankment protection	623	Right	4 no. of cross spurs

(continued)

Table 9.2 (continued)

Place	Types of artificial structures	Length (m)	Bank position	Remarks
Totapara	Cross spurs with linear embankment protection	763	Right	5 no. of cross spurs
Malpara	Linear embankment protection	631	Left	Bouldering along the bank
Sonakania	Linear embankment protection	3676	Left	5 no. of cross spurs
	Fair-weather bridges	230 (average)	Left	5 no. bridges made up to a sand bar in the right bank (mainly used for sand transportation)
Mahulpuli	Linear embankment protection	882	Right	4 no. of cross spurs
Gurudaspur	Fair-weather bridges	342	Across the river	Used for public transport
Jaleswar	Cross spurs with linear embankment protection	2744	Left	13 no. of cross spurs
	Fair-weather bridges	175 (each)	Left	One used mainly for sand transport and another for public transport
Mahulia	Cross spurs without linear embankment protection	1765	Right	15 no. of cross spurs
Rajghat	Cross spurs with linear embankment protection	325	Right (upstream)	1 no. of cross spur
		365	Right (downstream)	2 no. of cross spur
	Railway bridges	536 (north), 573 (south)	Across the river	Railway
	Highway bridges	609 (north), 582 (south)	Across the river	Roadway
Ramchandrapur	Cross spurs without linear embankment protection	415	Right	2 no. of cross spurs
Datapara	Cross spurs without linear embankment protection	376	Right	3 no. of cross spurs

(continued)

Table 9.2 (continued)

Place	Types of artificial structures	Length (m)	Bank position	Remarks
Kadarayan	Cross spurs without linear embankment protection	623	Right	4 no. of cross spurs
Sanroutpara	Cross spurs with linear embankment protection	674	Left	3 no. of cross spurs
Adimpur	Cross spurs with linear embankment protection	1614	Left	6 no. of cross spurs
	Fair-weather bridge	2207	Left	Made up to the sand bar in right bank mainly used for public transport
Jhaljhalia	Cross spurs with linear embankment protection	2938	Right	11 no. of cross spurs
Namkana	Cross spurs with linear embankment protection	1827	Left	2 no. of cross spurs
Parulia (Baliapal)	Cross spurs with linear embankment protection	2544	Right	11 no. of cross spurs
	Baliapal–Kamarda bridge	924	Across the river	Roadway
Aruhabarti	Cross spurs with linear embankment protection	1343	Left	8 no. of cross spurs
Athabatia	Cross spurs with linear embankment protection	3525	Right	21 no. of cross spurs
Rasalpur	Cross spurs with linear embankment protection	644	Left	3 no. of cross spurs
Patharghata	Cross spurs with linear embankment protection	2826	Left	8 no. of cross spurs
Bhusandeswar	Cross spurs with linear embankment protection	755	Left	3 no. of cross spurs
Kirtania	Linear embankment protection	1291	Left	Only boulder and basalt pitching

Source: Field survey records from December 2011 to March 2014.

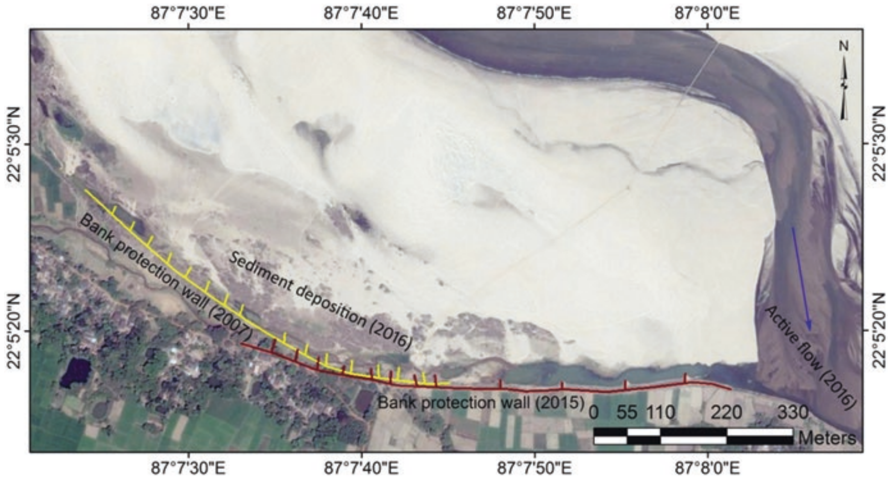

Fig. 9.6 Gradual downstream shifting of river active flow and energy concentration zone synchronizes with the positional shifting of the bank protection wall, and sediment deposition in the same position of earlier active flow concentration area at the right bank in Bhasraghat section

margin land within 9 years (Fig. 9.4). The excessive sand mining has been done even up to 2 m depth in compared with the natural sand layer of the riverbed exposing the boulder layer (Fig. 9.3a, b). The exposed boulders are also excavated by the local people (Fig. 9.3a). Therefore, the hydrodynamic behavior is changed in the deeper channel portion, which promoting the channel margin land erosion (Figs. 9.3a-b and 9.4). The channel margin land failure is initiating the formation of mud-balls after the dislocation of the muddy layer (Fig. 9.3b). Moreover, the intensive sand mining activities are being continued directly from the riverbed in the areas of Bhasraghat (Fig. 9.3c), Sonakania (Fig. 9.3d), and also in the different stretches (Fig. 9.2; Table 9.1). The intensive mining activities are leading to erosion of upstream point bars and the riverbed sediments that were deposited in the preceding flood events. The upstream eroded sediments are deposited in the downstream depression areas of the mining sites. Therefore, the sand mining activities have been continued throughout the year (except for the extreme flood period). All these actions have cumulative effects on the change of riverbed elevation, hydrodynamic behavior, and morphological features.

The erosion protection structures have both negative and positive impacts on the morphodynamic adjustment of the river course. The cross spurs with a linear embankment structure can arrest the sediment particles in the pocket section of the cross spurs (Fig. 9.5a) due to secondary flow turbulences. These arrested sediments help to stabilize the embankment and the erosional site may be altered as a depositional site (Fig. 9.6). Recently, a very low elevated and less extended fill terrace has been formed time in the same position of the earlier erosional sites at the right bank position in the Bhasraghat section (Fig. 9.6). However, the downstream section and the opposite bank position of the embankment are affected by severe erosion through the extreme turbo-mechanism of helicoidal flow. Therefore, bank failure has been associated in the opposite bank of the embankment (Fig. 9.5a) and also in the downstream section of the embankment (Fig. 9.6). The riverbed morphological features and riverbed elevation have also been modified after the active flow diversion for the construction of a fair-weather bridge across the river course (Fig. 9.5c-d).

3.1.3 Ecological Impact

Different types of species (flora and fauna) are living on the distinctive morphological setup of the riverbed depending on the seasonal fluctuation of flow. The specific ecological flow is required for the sustainability of the species. The unreal anthropogenic activities create a harsh impact on the natural ecosystem of the riverbed. Due to the intensive sand mining activities, some depression areas or pits have been formed. During the reduction of flow regime, those depression areas have been chocked and converted as semipermanent wetlands with the growth of some marshy floral species (Fig. 9.7c, d). During the construction of fair-weather bridge across the river stretch, the natural river flow has been obstructed and diverted, which emphasized to the growth of algal mat in the stagnant river water (Fig. 9.7a, b).

Fig. 9.7 Ecological impacts of sand mining from riverbed and flow diversion in the different river stretch. (**a**) Formation of ponding situation in the river course due to flow diversion, which occupied by the algal colony in degraded water and (**b**) thick algal mat on the riverbed near Bhasraghat; (**c**) growth of the algal colony in the degraded mining pit near Jaleswar; and (**d**) the degraded mining pit has been converted into marshy land near Rohini

Moreover, the wetland species have grown in the backwater cum ponding situation after embankment construction (Fig. 9.5b).

3.1.4 Impact on Socioeconomic Status

Sand mining and different structural measures have some positive socioeconomic impacts on the river fringed societies. The government, local dwellers, and other entrepreneurs earn a significant level of income from the sand mining activities. As per the field survey in 2011, the government collected 15,000 rupees from each mining site (khadan) for 5 years (Table 9.3). The government earned 390,000 rupees from 26 khadans within the studied river section (Table 9.1). Moreover, a large number of local people sustained their livelihood through sand mining and track loading activities after earning 250 rupees per day (Table 9.3). The details of economic turnover from the sand mining activities have been estimated in Table 9.3. Furthermore, the huge economic benefit is associated with the construction of structural measures in the different through minimizing the agricultural land erosion, reduction of crop failure from flood inundation, and sand deposition in the agricultural field. Different kind of bridges helps to easy transport and communication among both sides of the riparian people. The local people improve their socioeconomic status after involvement in the different constructional activities.

Table 9.3 Estimated turnover of the sand mining activities from the Subarnarekha riverbed at the Gopiballavpur section

Parameters	The right bank of Subarnarekha (near Nayabasan bar)	The left bank of Subarnarekha (near Gopiballavpur Bridge)
Sand mining area (km^2)	0.90	0.95
No of mining stations (khadan)	5	5
Mode of mining	Machine	Machine
Avg. mining intensity	300 tract/day (300 cft/track)	225 track/day (300 cft/track)
No. of labor	300	200
Avg. daily earning by a labor (Rs.)	250	
Govt. earns from each khadan for 5 years (Rs.)	75,000 (15,000 × 5 khadan)	75,000 (15,000 × 5 khadan)
Rate of sand of each track (Rs.)	300 (for 300 cft); 400 (for 400 cft)	
Labor charge for loading of each track (Rs.)	800 (for 300 cft); 1100 (for 400 cft)	

Source: Field survey on December 10th, 2011.

3.2 Loss of Channel Capacity

The channel capacity indicates the weighted perimeter of the channel, which is associated with the cross-sectional area, flow regime, and bank height in a particular river stretch. The cross-sectional area varies according to the morphological diversities coupled with riverbed elevation. The morphological features of extended sand bodies in the active channel bed, sand bars, mid-channel bars, point bars, pools, and cut-fill terraces are the major controlling factors of the channel capacity. The riverbed gradient is also playing an important role in the flowing of water and resultant depositional landform formation. The depositional landforms naturally reduce the channel capacity, whereas, the channel capacity increases with the sediment erosion from the riverbed and channel margin areas. The agricultural activities in the riverbed and bank margin deposited land also influences the changes of channel capacity. In the Rohini section, the mid-channel bar has been altered into agricultural land. This place is well known as the 'Kodopal Eco-nest and Biological Fruit Basket' (Jana 2017). Also, in the different river reaches, the fill terraces, and point bars have been converted into agricultural land. People are cultivating crops over the immature landforms of point bar, fill terrace, and mid-channel bar, and the natural stability of the landforms has been reduced after ploughing the immature land surface for crop cultivation. The loose soil particles are easily removed from the agricultural land by the rainwater and floodwater and deposited over the riverbed, which is also reducing the channel capacity. Besides, the channel margin embankment areas have been used for agricultural land and settlement formation after leveling the sloppy surface by cutting the upper portion of the embankment. The natural stability of the embankment is reduced and it eroded down. Generally, the agricultural wastes have been dumped in the river margin areas by the farmers, which also leads to the reduction of channel capacity. The channel capacity has been diminished after the construction of the bridge protection embankment structure in the left bank at Gopiballavpur (Fig. 9.5b).

3.3 Recommended Management Strategies

Concerning the overall analysis of anthropogenic as well as natural activities within the present study section, the following strategies can be adopted for minimizing the erosional effects related to sand mining and construction of structural measures in the different reaches of the Subarnarekha River.

1. Sand and gravel acquiring can be allowed only in the specific areas of the selective sites on the riverbed under the proper surveillance.
2. No machine should be introduced for sand mining activities.
3. The proper guideline should be implemented regarding the period (permitted and restricted months for mining) and intensities of san mining of a year.

4. Depending on the volume of sand deposition the permission should be given to sand mining. Otherwise, sand acquiring has been completely ceased for that session.
5. No mining activities should be permitted in the erosive riverbed and its marginal areas.
6. The degraded and the depression mining places should be artificially filled up by the miners (owner) after the unscientific and illegal mining.
7. Artificial way or fair-weather bridge should not be constructed over the riverbed for the sand transport by trucks.
8. Need to control activities of illegal sand mining and sand transport.
9. Regular observation by the government authority should be required for the mining-related activities and taxes.
10. Erosion protective embankment and cross spur construction may be allowed after the proper assessment of natural loss and social beneficial response both in the constructed site and its surrounding areas.
11. Agricultural activities should not be permitted in any landforms of the river course.

4 Conclusion

The naturally deposited sand has a great contribution for maintaining the balance between the energy-material interaction during and after the flood events every year. The extraction of deposited sand can be allowed to some extent depending on the natural setup of the channel. In most of the cases, people are extracting the sand from the river bed and the channel fringe areas without any care about the channel behavior. If any position of the mining place is gradually affected by the river erosion, then there should need immediate prevention and control of the sand mining activities. But, observing the economic benefit and dependency of the people on the sand mining activities, it can be done up to a certain limit, maintaining the sustainable level of the natural setup of the channel. The bank erosion nature can be control through the installation of embankment structures only in the particular site, but the opposite bank and the downstream reach of the embankment are harshly affected by the erosion associated with the changing hydrodynamic behavior. Therefore, the study of hydrodynamic behavior is essential before the construction of erosion prevention structures and various bridges along and across the river course. The intensive sand mining activities need to control in the entire studied section of the Subarnarekha River. But, as a huge volume of sand is deposited within the different channel reaches, the limited volume of sand mining can be permitted after observing the hydrodynamic behavior of the channel related to erosion. There have an urgent need to maintain the natural hydro-morphodynamic behavior of the studied river section not only for the sustainability of the river itself but also to sustain the riparian dwellers through controlling the excessive population growth and socioeconomic demands.

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Chapter 10
Perturbation of the Health of the Riverine Ecosystem and its Impact on the Biogeochemical, Ecological, and Molecular Perspectives

Ritwik Patra, Saubhik Mitra, and Suprabhat Mukherjee

Abstract "Pollution" is probably the deadliest "word" of the current generation which is steadily evidencing its identity in the lithosphere, atmosphere, and biosphere. The concurrent progress of human civilization and various industries continuously added stimulation in polluting the environment. In this context, water pollution especially riverine pollution has exerted enormous noxious influence over the biosphere due to its connection with the diverse ecologically related living organism comprising microbes, plants, and animals. Toxicants belonging to phenolics, organometallic compounds, and heavy metals originating from industrial effluents and wastes are the prime contributors to pollution. Increasing the abundance of nano-toxicants is slowly emerging as an added complication. Although all living organisms are equipped with a distinct detoxification system, but pollution beyond the tolerance level results in damage and death. Depletion of beneficial microbes, induction of microbial resistance contrary to heavy metals, and organic compounds leading to the emergence of altered pathogenicity are the key phenomena associated with the microbial communities of the river. Similarly, loss of growth and productivity leading to death are the major concerns for the river-associated flora. Animals are one of the major members of the riverine ecosystem and several physiological damages, namely, loss of growth, reproductive defects, poor immunity, and death are documented in several studies. Animals and humans are also affected by the pollutants through the dietary consumption of the aquatic animals contaminated with the pollutants. Intriguingly, pollution-induced damages in the riverine ecosystem are also concerns for the other ecological niches, especially those that are con-

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nected to the riverine ecosystem either directly or across the food chain. Riverine pollutants have been shown to induce oxidative damages, membrane damage, chromosomal aberrations, and induction of apoptosis. Alarmingly, increasing occurrences of different forms of cancers are also found to be connected with water pollution. Several remediation strategies have been adopted to counteract the noxious pollutants; however, limited efficacy of the approaches, continuous addition of the pollutants, and lack of public awareness have posed obstacles in restoring the health of the river. In this chapter, the toxicology of riverine pollution caused by organic and inorganic contaminants and pollutants-induced perturbations of the riverine ecosystem at a cellular and molecular level, as well as their management, have been discussed based on existing and upcoming approaches.

Keywords Riverine ecosystem · Riverine pollution · Phenolics · Organometallic compounds Heavy metals · Toxicology · Apoptosis · Cancer

1 Introduction

"Pollution" is probably the deadliest "word" of the current generation which is steadily evidencing its identity in the lithosphere, atmosphere, and biosphere. The concurrent progress of human civilization and various industries continuously added stimulation in polluting the environment. In this context, water pollution especially riverine pollution has exerted enormous noxious influence over the biosphere due to its connection with the diverse ecologically related living organism comprising microbes, plants, and animals.

Toxicants belonging to phenolics, organometallic compounds, and heavy metals originating from industrial effluents and wastes are the prime contributors to the pollution (Choi et al. 2004; Faroon et al. 2003; Martin and Griswold 2009). With the constant progress of civilization, industrial growth, and urbanization, the riverine ecosystem is being continually polluted day by day through the contamination of heavy metals and organic toxicants resulting in the damage and destruction of the aquatic flora and fauna (Isiuku and Ebere 2019). Organic pollutants in the riverine ecosystem include a wide range of xenobiotic compounds such as phenol and phenolic compounds, organometallic compounds including organochlorine pesticides and organophosphate pesticides, various herbicides and fungicides, hexachlorocyclohexane (HCHs), dichlorodiphenyltrichloroethane compounds (DDTs), and polychlorinated biphenyls (PCBs) (Deribe et al. 2011; Kannan et al. 2005; Pan et al. 2017). They enter the water bodies through discharge from industries, household wastewater, and sewage along with the flooding of agricultural lands. It can readily absorb within the system of a living organism, and because of its high persistence within the ecosystem, it gets easily accumulated and causes toxicity (Association 2017; Faroon et al. 2003; Tchounwou et al. 2012). The consequences of the toxicity lead to oxidative stress with the aquatic organism that further disrupts the antioxidative system, generates ROS causing damage in DNA, apoptosis, and even carcinogenesis (Goswami et al. 2015; Tchounwou et al. 2012). In a riverine ecosystem, waterborne heavy metals are easily taken up by the aquatic flora and fauna and are strongly bound with proteins resulting in accumulation within the tissue. Heavy metals are continuously discharged into the water bodies from natural and human sources which include industrial and domestic waste consisting of discharge of sewage, mining waste, farming pesticides and chemicals, and electronic waste (Del Ramo et al. 1987; Martin and Griswold 2009). Moreover, it can easily be coagulated within the water to get absorbed and bioaccumulated in the aquatic ecosystem that may be both helpful and lethal. Although in organisms some metals like iron, copper, manganese, zinc, and so on are necessary for their metabolic process, some are marked as hazardous due to their high toxicity including cadmium, mercury, lead, chromium, arsenic, and nickel. The persistence of heavy metals in the environment at a higher level is hazardous for both the ecosystem and human health (Isiuku and Ebere 2019; Jaishankar et al. 2014). The main consequences lead to bio-accumulation and bio-concentration across the food web leading to a surge in the concentration of the contaminants in the organisms over a while, resulting in toxicity and magnification through the food chain (Choi et al. 2004). Increasing the abundance of nanotoxicants is slowly emerging as an added complication.

Although all living organisms are equipped with a distinct detoxification system, pollution beyond the tolerance level results in damage and death. Depletion of beneficial microbes, induction of microbial resistance against toxicants, and the emergence of altered pathogenicity are the key phenomena associated with the microbial communities of the river (Bhat et al. 2020; Choi et al. 2004; Goswami et al. 2011). Similarly, loss of growth and productivity leading to death are the major concerns for the river-associated flora (Mukherjee et al. 2013; Urbaniak et al. 2019). Biological communities are a complex network of several components that include living and non-living entities. The network is complete with even more complex interactions between its components. The behavior of living organisms is one such thread that drives the community structure on a large scale (Scott and Sloman 2004). Predator–prey interaction keeps a community stable. A predator controls the population of its prey, thus giving a chance to further lower strata of the food pyramid to flourish. If the predatory behavior of that animal alters, the whole community suffers a misbalance, a top-down effect (Schmitz 2007). Similarly, if the predation-evading behavior of prey is altered to be a bolder one, the population will be exposed to a higher rate of hunting and eventually perish. In wild, only fittest survives, and so, less active, weak individuals are always at a risk.

Animals are one of the major members of the riverine ecosystem, and several physiological damages, namely, loss of growth, reproductive defects, poor immunity, and death are documented in several studies. Animals and humans are also affected by the pollutants through the dietary consumption of the aquatic animals contaminated with the pollutants. Intriguingly, pollution-induced damages in the riverine ecosystem are also concerns for the other ecological niches, especially those that are connected to the riverine ecosystem through the food chain or directly to the riverine water. The induction of microplastics into the aquatic system causes a serious issue in this present time (Ghosh et al. 2021). Riverine pollutants including both organic and inorganic contaminants have been shown to induce oxidative damages, membrane damage, chromosomal aberrations, and induction of apoptosis. Alarmingly, increasing occurrences of different forms of cancers are also found to be connected with water pollution.

Several remediation strategies have been adopted to counteract the noxious pollutants; however, limited efficacy of the approaches, continuous addition of the pollutants, and lack of public awareness have posed obstacles in restoring the health of the river. In this chapter, the toxicology of riverine pollution caused by organic and inorganic contaminants and pollutants-induced perturbations of the riverine ecosystem at cellular and molecular levels, as well as their management, have been discussed based on existing and upcoming approaches.

2 Toxicology of Riverine Pollution: Impact on the Microbial Communities, Plants, and Animals

Riverine pollution resulted due to the addition of various organic and inorganic contaminants through the expulsion of industrial wastewater and effluents, wastes of textiles and tanneries, nano-toxicants from industrial and household waste, and so on. As we have discussed in the introduction that riverine pollution does interfere with the health of the ecosystems and produces detrimental outcomes, the major organic pollutants causing perturbation of the riverine ecosystem have been discussed subsequently.

2.1 Effect of Organic Compounds on the Health of the Riverine Ecosystem

2.1.1 Phenol and Phenolic Compounds

Phenol and its associated phenolic compounds are the shared pollutants released from various industries that involve high-temperature coal conversion; petroleum purification; manufacturing of steel and metals products, resins and dyes, wood, and paper pulp, agrochemicals products; and pharmaceuticals industries (Holcombe et al. 1982). Chlorophenol and its derivatives are ubiquitous toxicant intermediates that are by-products in manufacturing agricultural pesticides, pharmaceuticals products, and dye industries (Igbinosa et al. 2013). These compounds can acts as electrophilic metabolites leading to bioaccumulation, causing acute to chronic toxicity, various histopathological changes resulting in mutation, and generation of cancers (Choi et al. 2004; Zhang et al. 2004). The International Agency for research on cancers characterized chlorophenols as Pentachlorophenol (PCP), 2,3,4,6-tetrachlorophenol (2,3,4,6-TeCP), various forms of trichlorophenol (TCPs) derivatives, and 2,4-dichlorophenol (2,4-DCP) as the toxic form (Igbinosa et al.

2013). Pentachlorophenol (PCP) and tetra chlorophenol (TeCP) and their sodium salts are on the priority list of the environmental toxin of the U.S. Environmental Protection Agency and the European Union (Igbinosa et al. 2013). Exposure of sodium pentachloro phenate (Na-PCP) in steelhead trout (*Salmo gairdneri*) increases the rate of oxygen consumption and decreases yolk utilization ability resulting in reduced growth (Hodson and Blunt 2006). Oxidative DNA base damage leads to the increase in reactive hydroxyl radical converted from hydrogen peroxide by the transition of Fe^{2+} or Cu^{2+}ions (Zhang et al. 2004). At low concentrations, pentachlorophenol may induce structural anomalies in chromosomes causing aberrations, and some para-substituted phenols like 4-chlorophenol were converted to hydroquinone using cytochrome P_{450}2E1(CYP2E1) (Igbinosa et al. 2013; Michałowicz and Majsterek 2010). In zebrafish, 2,4,5-trichlorophenol (2,4,5-TCP) leads to liver carcinogenesis by inducing point mutations of the p53 gene (Zhang et al. 2018). 2,4-dichlorophenol (2,4-DCP) acts as an endocrine disruptor (EDCs) and damages the antioxidative system in many freshwater fish like *Carassius auratus* (Zhang et al. 2004).

Alkylphenols with butyl and propyl substitutions are used as industrial products including oil demulsifiers, oil additives, antioxidants, plasticizers, and phenolic resins (Choi et al. 2004; Kang et al. 2007). Alkylphenol includes 3-tert-butyl-phenol, 2-isopropyl phenol, 3-isopropyl phenol, and 4-isopropyl phenol that causes acute to chronic toxicity starting from aquatic microbes, invertebrates, to higher fishes (Choi et al. 2004). The bacterium *Vibrio fischeri* was the most susceptible up to three times more than species of higher levels to the alkylphenols, particularly 4-isopropyl, and in the case of Ceriodaphnia it is sensitive and toxic at an EC50 value of 10.1 mg/L (Choi et al. 2004). Nonylphenol and its epoxide derivatives act as an endocrine disruptor and also damage the oxidative system in zebrafish releasing reactive oxygen species (ROS) (Caballero-Gallardo et al. 2016). Chronic toxicity to nonylphenol leads to reproductive changes and damages. In *Labeo bata,* it results in the appearance of ovotestis.

Bisphenol A (BPA) is the most toxic phenolic compound utilized in the manufacture of phenol resins, polyacrylates, polycarbonate plastics, and polyesters. It has greater reactivity with chlorine, and these chlorinated derivatives are cytotoxic and show potent estrogenic activity in both males and females of the midge *Chironomus riparius,* causing the delay of the emergence of the second generation (Kang et al. 2007). There is evidence of a reduction in the sperm density on exposure to BPA in brown trout (*Salmo trutta*) at concentrations of 1.75–2.4 μg/L (Lahnsteiner et al. 2005). Effects of bisphenol at a concentration of >100 μg/L at the initial stage of fish result in embryonic deformities. It also induces chromosomal damage, depletion of erythrocytes, and reduction of micronuclei frequency consequences leading to lower the number of eggs and hatchings in *Scophthalmus maximus* (Bolognesi et al. 2006). In Zebrafish (*Danio rerio*), exposure to bisphenol for 5–7 days shows vitellogenin response. It can induce reproductive changes of testis–ova in Medaka (*Oryzias latipes*) and malformation of tail flexure in black-spotted pond frog (*Rana nigromaculata*) on exposure to 200 μg/L for 45 days (Kang et al. 2007).

2.1.2 Organophosphorus Compounds and their Derivatives

The organophosphates and carbamates are highly toxic synthetic insecticides having the potential to damage ecosystem function on their toxicity, exposure time, dose rate, and persistence. The median lethal concentration (LC50) is categorized as either acute or chronic toxicity (Mustafa et al. 2014). In bluegill (*Lepomis macrochirus)*, LC50 of chlorfevinifos, diazinon, and profenofos were determined as 2.9 ppm, 2.5 ppm, and 300 ppb, respectively. 2.8 mg/L dose of carbofuran in *Catla catla* causes 100% mortality within 96 hours of exposure and also shows various consequences that lead to asphyxia, restlessness, equilibrium disbalance, and erratic swimming (Mustafa et al. 2014). A comprehensive analysis by Mukherjee et al. (2015) showed the molecular mechanism of damages induced by the organophosphate insecticide and phosphamidon in a rat model. Phosphamidon was found to induce lipid peroxidation of the cell membrane and the generation of reactive oxygen species (ROS) that collectively resulted in disruption of the redox homeostasis (Mukherjee et al. 2015). Moreover, ROS was shown to signal the activation of the apoptotic pathway resulting death of the cells as well as the experimental animal (Mukherjee et al. 2015).

The effect of exposure of methyl parathion on freshwater characid fish (*Brycon cephalus*) and *Daphnia laevis* induce oxidative stress resulting in the production of ROS and variations in antioxidants or free oxygen radicals scavenging enzyme systems and results in a surge of glutathione peroxidase (GP), catalase (CAT), glutathione S-transferase (GST), superoxide dismutase (SOD), and reduced glutathione (GSH) (Monteiro et al. 2006). These ROS like highly reactive superoxide anion radical ($O^{2\bullet-}$), hydrogen peroxide (H_2O_2), and hydroxyl radical (•OH) are susceptible to biological macromolecules causing lipid peroxidation (LPO), protein oxidation, and DNA damage (Monteiro et al. 2006).

Diazinon is widely used as pest control in agriculture and domestic use, and on contamination with the aquatic environment, it is accumulated in the gills of fishes (Al-Otaibi et al. 2019). Another physiological response reflects through the hematological parameters like a decrease in hemoglobin, blood cell counts, glycemia, and change in ion concentrations (Al-Otaibi et al. 2019). Diazinon is moderately toxic to catfish (*Clarias gariepinus*), and diazinon shows a moderate level of toxicity and at lower doses can alter the biochemical and hematological profile resulting in a significant increase in GOT and GPT susceptibility causing an elevated level of blood glucose (Al-Otaibi et al. 2019).

Ethoprophos shows the neurotoxic effect on fish that leads to a slow escape response to a predator attack. Phosmet leads to acute to chronic toxicity and bioaccumulation in various fishes like *Lepomis macrochirus*, *Oncorhyncus mykiss*, and *Gammarus fasciatus*. An inhibitory effect on AChE response is observed in Crucian carp (*Carassius carassius)* and *Channa punctatus* on exposure to triazophos for 5–7 days (Singh et al. 2018).

2.1.3 Organochlorines and Polychlorinated Derivatives

Organochlorine is made up of carbon, hydrogen, and chlorine and is considered a global pollutant having higher bio-accumulative and toxicity (Chopra et al. 2011). The highest level of damage due to organochlorines is seen in the Irrawaddy dolphin (*Orcaella brevirostris*) found all along with Southeast Asia, Bay of Bengal, Indo-Malay Archipelago, to northern Australia (Kannan et al. 2005). It is categorized as the world's most seriously endangered species and also has a threat of extinction. Indus river dolphin (*Platanista minor*) in Pakistan and Yangtze river dolphin (*Lipotes vexillifer*) found in China are already close to extinction (Kannan et al. 2005). DDTs and their metabolites are the principal contaminants, with the highest concentration recorded of 10,000 ng/g of lipid weight in blubber (Kannan et al. 2005). These lead to starvation or disease-causing mobilization of lipids from blubber to kidney or liver.

Lindane is a group of γ-hexachlorocyclohexane compounds that is accumulated within the aquatic organism directly or via the food chain (Ding et al. 2015; Kannan et al. 2005). It decreases the growth of the juvenile of freshwater amphipod *(Gammarus pulex*) by damaging the hepatopancreatic tissue of the digestive system (Chopra et al. 2011).

PCBs are synthetic chlorinated organic compounds, used as lubricants, coolants, and flame retardants in various industries (Faroon et al. 2003). The main source of contamination in the river is due to anthropogenic activities that include farming, intensification of erosion, discharge of unregulated wastewater from factories as well as accelerated transport of micro-pollutants from various sources (Kannan et al. 2001; Walters et al. 2011). PCDDs, PCDFs, and dl-PCBs have extremely low water solubility, so they can easily bind with the organic and mineral particles across the aquatic environment resulting in sedimentation and contamination, which gradually leads to accumulation and persistence within the ecosystem (Urbaniak et al. 2019). At high concentrations, it causes a nuisance to the health of benthic organisms, including fishes (Faroon et al. 2003). PCBs are majorly found in Trenton Channel from the Detroit River and show a high impact on Brown bullheads (*Ameiurus nebulosus*) (Kannan et al. 2001). It leads to various structural anomalies causing lesions on skin and lips, various physiological effects resulting in induction of cytochrome P450 enzyme, development of liver tumors, and carcinogenesis. It also affects the liver fat and muscle of *Thunnus thynnus*. PCB 52 and PCB 77 lead to alteration of oxidative metabolism system in various fishes (Kannan et al. 2001; Urbaniak et al. 2019). The molecular mechanism of induction of the toxicity in the aquatic animals by different organic pollutants, especially belonging to PCB, is summarized in Fig. 10.1.

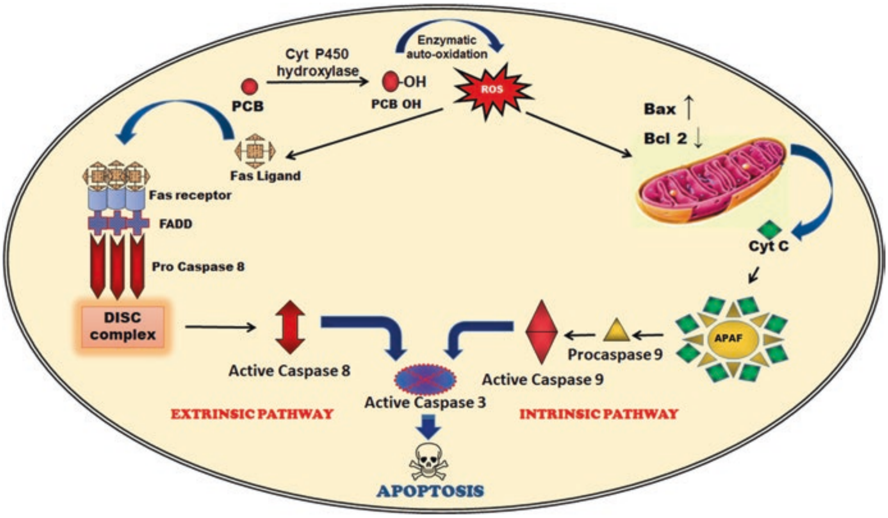

Fig. 10.1 Mechanistic insights of cytotoxicity induced by the PCB**.** PCB is generally transformed into a more toxic form after entering into the animal body and induces oxidative stress. Reactive oxygen species formed in the cells further signal activation of apoptotic pathways to result in cell death

2.1.4 Pyrethroids

Pyrethroids are a group of insecticides that even at a low dose of the application show high bioactivity (Antwi and Reddy 2015). It is water-insoluble and has a strong absorptive ability. It can be classified into two major classes, Type I and Type II, based on their potentiality of toxic nature and extends of damage. The structural difference between them is the presence of an alpha-cyano group in type I (Antwi and Reddy 2015; Prusty et al. 2015; Werner and Moran 2008). Type I pyrethroids include permethrin and cismethrin and is highly neurotoxic, and interfere with the Na^+/K^+ channel in even closed state causing the presynaptic repetitive release (Prusty et al. 2015). Type II pyrethroids such as deltamethrin, esfenvalerate, cypermethrin, and bifenthrin can affect the Na^+/K^+ channel along with Cl^- and Ca^{2+} channels (Prusty et al., 2015). Pyrethroids act as an agonist and induce the synthesis of acetylcholinesterase and can inhibit ATPase activity (Antwi and Reddy 2015; Prusty et al. 2015). It is an endocrine disruptor resulting in decrease of progesterone and estradiol biosynthesis in various aquatic vertebrates (Antwi and Reddy 2015; Prusty et al. 2015). The efficiency of toxicity increases by the breaking down of pyrethroids inflicting various estrogenic and anti-androgenic effects in females and males, respectively.

Deltamethrin can easily bind to the slit particles within the riverine ecosystem causing bioaccumulation (Köprücü and Aydın 2004; Singh et al. 2018). It is lipophilic resulting in smooth absorption within the gills and tissues of fishes inflicting histopathological modification within gills and liver of fish *Oreochromis niloticus.*

In zebrafish, *Danio rerio* disrupts the calcium and phosphate homeostasis leading to various physiological and reproductive changes resulting in an increase in atretic oocytes (Prusty et al. 2015). Short-term exposure to deltamethrin leads to an imbalance of electrolytic potential causing hypocalcemia and hypophosphatemia leading to renal dysfunction in *Heteropneustes fossilis* (freshwater catfish) (Prusty et al. 2015).

Cypermethrin shows more efficiency of toxicity at the early stages of development. In *Labeo rohita*, it causes histopathological alteration of liver and kidney causing swelling of hepatocyte, hyperplasia, and coagulative necrosis (Prusty et al. 2015). It inhibits the AChE thus elevating the acetylcholine activity with the gills and brain and liver of various fishes like *Channa punctatus and Cyprinus carpio* (Prusty et al. 2015). Consequences of these lead to manifestation in CNS, body equilibrium, and show many behavioral changes like erratic swimming.

2.1.5 Herbicides

The main source of contamination of herbicides in the aquatic ecosystem is through the discharge of excess water from agricultural fields and urban areas. It mainly targets the aquatic herbivore thus creating a disturbance to the food chain (Solomon et al. 2008). Atrazine primarily affects the phototropic microorganism and an increase in algal biomass (Cui et al. 2019). This leads to a rapid decrease in the number of phytoplanktons causing scarcity of foods. In freshwater fish (*Channa punctatus*) and common carp, it damages the antioxidative system leading to the generation of ROS and lipid peroxidation (Solomon et al. 2008). It also exhibits various immune dysfunction causing inflammation and disruption of glycometabolism in lymphocytes that lead to necroptosis (Cui et al. 2019).

Glyphosate and its derivatives are non-specific systemic herbicides. It targeted the aquatic plants and algae causing interruption to phytoplankton and periphyton communities (Pérez et al. 2011). It inhibits the growth of cyanobacteria and sensitivity to other aquatic protozoans. The snail *Pseudosuccinea columella* under very low concentration shows an increment in third-generation growth causing inhibition of eggs hatching, polyembryony, and birth abnormalities (Pérez et al. 2011). In certain riverine tadpoles of *Rana cascadae*, it shows early metamorphosis that leads to an underdeveloped adult structure (Pérez et al. 2011). Glyphosate shows both direct and indirect effects on the fish population. It reduces the bioavailability of food along with high toxicity that leads to juvenile mortality in *Oncorhynchus mykiss* (Chen et al. 2004). The molecular mechanism of induction of the toxicity in the aquatic animals and microbes by different insecticides and pesticides is summarized in Fig. 10.2.

A generalized summary of the major organic pollutants, their target organisms, and the mechanism of damaging the riverine ecosystem is given in Table 10.1.

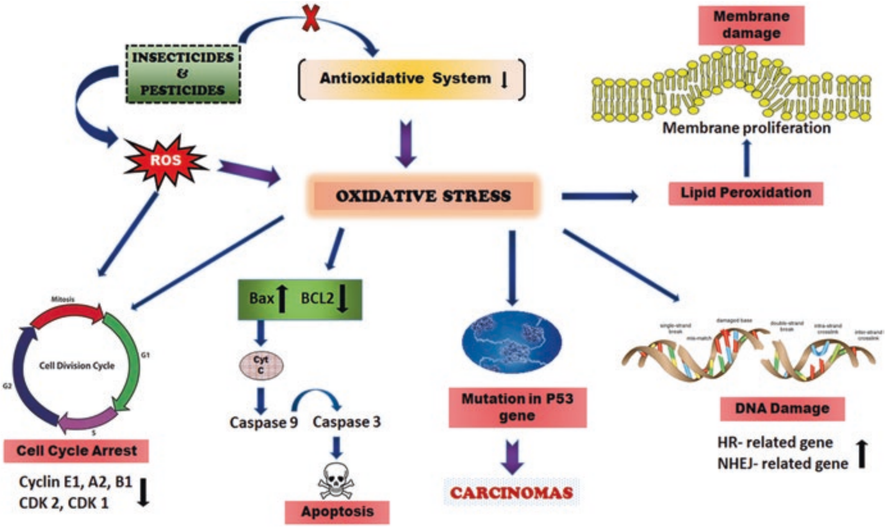

Fig. 10.2 Mechanism of induction of toxic alterations and damages by insecticides and pesticides. Pesticides and insecticides majorly trigger disruption of redox homeostasis and induction of reactive oxygen species. Reactive oxygen species results in the arrest of cell cycle progression, activation of apoptotic pathways to result in cell death, and importantly deactivation of tumor suppressor gene

2.2 *Effects of Heavy Metals on the Health of the Riverine Ecosystem*

2.2.1 Cadmium

Cadmium (Cd) is an extremely poisonous heavy metal and is listed seventh in the ATSDR (Agency for Toxic Substances and Disease Registry) ranking (Table 10.2). It is used extensively in mining, science operations, electroplating industries, producing vinyl plastics, and creating electrical instruments and metallic and plastic pipes (Tchounwou 2012). When compared to alternative metals, the water solubility of Cd and its derivative are high causing tremendous bioavailability and bioaccumulation properties. Aquatic organisms usually uptake Cd as a free ionic Cd(II) type (Ferard et al. 1983). Cd combines with chlorides and forms a less accessible molecule in marine than the virulent form in freshwater (Qu et al. 2013). The bivalve *Caelatura companyoi* and the snail *Cleopatra bulimoides* can accumulate Cd in their soft tissue and are therefore used as a bioindicator for determining the extent of toxicity within the river (Moloukhia and Sleem 2011). Consequences of Cd toxicity in fishes lead to various physiological changes like bioaccumulation in liver and kidney, reproductive dysfunction, and immune dysfunction through induction of apoptosis in macrophages and T cells (Kitana and Callard 2008; Yuan et al. 2017). Skeletal deformities in fish due to high accumulation of Cd end up in the loss of the ability in searching food and avoiding predators causing sublethal effects

Table 10.1 Different organic compounds pollute and disrupt the riverine ecosystem

Types of pollutant	Name of the compound	Chemical class/ nature	Organisms affected	Mode of action and extent of damage	References
Phenols and Phenolics	2,4-dichlorophenol (2,4-DCP)	Chlorophenol	Freshwater fish (*Carassius auratus*)	Antioxidant system damager and endocrine disruptor	Zhang et al. (2004)
	2,4,5-trichlorophenol (2,4,5-TCP)		Zebrafish	Results in point mutations in p53 gene in liver causing carcinogenesis	Yin et al. (2008)
	Pentachlorophenol(PCP)		Common guppy (*Lebistes reticulatus*), steelhead trout (*Salmo gairdneri*)	Results in a reduction of growth, increased rate of oxygen consumption, decreased yolk-utilization, and DNA damage	Igbinosa et al. (2013)
	Pentachlorophenyl laurate (PCPL)		Bluegill *(Lepomis macrochirus),* largemouth bass *(Micropterus salmoides)*	Bioaccumulation and toxicity	Hodson and Blunt (2006)
	Catechol	Simple phenols	*Escherichia coli*	Damage of plasma membrane	Schweigert et al. (2001)
	2-Isopropylphenol,4-Isopropylphenol, Nonylphenol (NP) and its epoxide derivatives	Alkyl-phenol	Ceriodaphnia, *Vibrio fischeri, Labeo bata, and* zebrafish	Sensitivity and toxicity, reproductive changes/damages; the appearance of ovotestis; endocrine disruptor and oxidative damage	Choi et al. (2004), Caballero-Gallardo et al. (2016)
	Bisphenol A (BPA)		Zebrafish, *Scophthalmus maximus*	Endocrine disruption, peripubertal mammary gland development; embryo affected; chromosomal damage, reduction in the amount of egg formation and hatchings	Kang et al. (2007), Lahnsteiner et al. (2005) Bolognesi et al. (2006)

(continued)

Table 10.1 (continued)

Types of pollutant	Name of the compound	Chemical class/ nature	Organisms affected	Mode of action and extent of damage	References
Pyrethroids	Deltamethrin	Type 2 pyrethroid	*Fish (Oreochromis niloticus*), Zebrafish (*Brachydanio rerio)*	Causes histopathological alterations in the gills and liver; calcium and phosphate homeostasis, ill effect on the reproductive system and increases atretic oocytes	Prusty et al. (2015)
	Permethrin	Type 1 pyrethroid	*Diptera* (*Chironomus riparius*), Shrimps (*Caridina africana)*	Decreased larval density and adult emergence. Reduction in population density	Antwi and Reddy (2015)
	Cypermethrin	Type 2 pyrethroid	Common carp (*Cyprinus carpio* L.), Caspian roach (*Rutilus rutiluscaspicus),* silver carp (*Hypophthalmicthys molitrix*), African catfish (*Clarias gariepinus*)	Larval mortality, respiratory stress, hyperplasia, hepatic lesion, and coagulative necrosis	Prusty et al. (2015)
	Esfenvalerate	Type 2 synthetic pyrethroid	Water boatmen (Corixidae), fish (*Sarcamento splittail*), *Melanotaenia fluviatilis*	Results in vacuolar degeneration and necrosis across the liver, decrease fecundity rate, and inhibits egg hatching	Prusty et al. (2015)
	Fenvalerate		C. *mrigala, C. batrachus, Tilapia mossambica*	Alteration of tissues of gill, kidney, and liver, liver necrosis, vacuolization, and membrane damage	Prusty et al. (2015)
	Monocrotophos		Freshwater bivalve (*Lamellidens marginalis), fish (Tilapia mossambica, Labeo rohita)*	Inhibits acetylcholinesterase, increases excitability in neurons, and histopathological changes in the gill and liver	Tamizhazhagan and Pugazhendy (2016)

Types of pollutant	Name of the compound	Chemical class/ nature	Organisms affected	Mode of action and extent of damage	References
Organophosphate	Parathion		Freshwater characid fish (*Brycon cephalus*), *Daphnia laevis*	Acute toxicity due to induction of oxidative stress	Monteiro et al. (2006)
	Chlorpyrifos (CPS)		*Daphnia magna,* Japanese medaka (*Oryzias latipes*)	Bioaccumulation, hyperactivity, and elevated anxiety	Qiu et al. (2017)
	Dichlorvos		Carp (*Cyprinus carpio* L.) and catfish (*Ictalurus nebulosus*)	Oxidative damage	Monteiro et al. (2006)
	Diazinon		Catfish (*Clarias gariepinus)*	Histopathological changes in liver, gill, and kidney tissue as well as hematological changes	Al-Otaibi et al. (2019)
	Phosmet		Bluegill sunfish (*Lepomis macrochirus)*, rainbow trout (*Oncorhyncus mykiss)*	Acute to chronic toxicity	Singh et al. (2018)
	Triazophos		Crucian carp (*Channa punctatus*)	Inhibitory effect on acetylcholine esterase (AChE) activity	Singh et al. (2018)

(continued)

Table 10.1 (continued)

Types of pollutant	Name of the compound	Chemical class/ nature	Organisms affected	Mode of action and extent of damage	References
Organo-chlorine	Lindane, γ-HCH	*Gamma*-hexachlorocyclohexane (γ-HCH)	Brown trout (*Salmo trutta*), water flea (*Daphnia magna*), freshwater amphipod (*Gammarus pulex*)	Acute toxicity, growth inhibition, and disruption of the architecture of hepatopancreatic tissue	Chopra et al. (2011), Kannan et al. (2005)
	p,p′-DDE; o,p′-DDT	DDT	Riverine dolphins	Bioaccumulation and toxicity in blubber	Kannan et al. (2005)
	PCBs like PCB52, PCB77	Polychlorinated compounds	Brown bullheads (*Ameiurus nebulosus*), river turtles	Disruption of oxidative metabolism, epidermallesion, increased cytochrome P450, liver carcinomas	Faroon et al. (2003), Urbaniak et al. (2019)
	Polychlorinated dibenzofurans (PCDFs)		*Thunnus thynnus*	Bioaccumulation in fat, liver, and muscles	Kannan et al. (2001)
Herbicides	Atrazine	Triazine class	Freshwater fish *(Channa puntatus)*, common carp	Lipid peroxidation and antioxidant dysfunction, inflammatory reaction, and dysregulation of glycogen metabolism in lymphocyte	Cui et al. (2019), Solomon et al. (2008)
	Glyphosate		Snail (*Pseudosuccinea columella), Rana cascadae, Oncorhynchus mykiss*	Growth inhibition, early metamorphosis, and juvenile mortality	Chen et al. (2004)

Table 10.2 Priority List for the heavy metals (as per ATSDR2019[a])

Heavy metals	ATSDR rank	Total point
Arsenic	1	1676
Lead	2	1531
Mercury	3	1458
Cadmium	7	1318
Cobalt	52	1011
Nickel	58	993
Zinc	75	913
Chromium	78	893
Lead-210	122	805

[a]Source: https://www.atsdr.cdc.gov/spl/index.html

more deadly. It additionally disrupts glutathione metabolism in *Oreochromis niloticus* (Eroglu et al. 2015). In humans, long-run exposure of Cd through consumption of contaminated water causes nephritic dysfunction, showing various symptoms of tubular proteinuria, kidney failure, and so on (Jaishankar et al. 2014; Tchounwou 2012). Inhaling higher levels of Cd from the contaminated water bodies or rivers can cause severe damage to the lungs inflicting Cd pneumonitis (Jaishankar et al. 2014). The main consequences are chest ache showing cough having foam and blood sputum and damage across the lung tissue lining leading to fluid accumulation (Tchounwou et al. 2012), (Sujata 2015). Bone defects are also associated with Cd and reported for disrupting calcium metabolism, resulting in osteomalacia, osteoporosis, and spontaneous fractures. In adverse conditions, Cd also enhances blood pressure and increases the risk of myocardial dysfunctions (Jaishankar et al. 2014; Tchounwou 2012).

2.2.2 Lead

Lead (Pb) is also considered as the frontier hazardous heavy metal. Pb is especially utilized in the manufacturing of batteries, cosmetics, metal products, namely, ammunitions, solders, pipes, and so on (Jaishankar et al. 2014; Martin and Griswold 2009). The primary sources of exposure are Pb-based paints, industrial emissions, cosmetics, household dust, gasoline, contaminated soil, and toys (Jaishankar et al. 2014). Within the aquatic ecosystem, Pb occurs in various forms such as mobile bioavailable ionic form, organic complexes forms with dissolved humus showing limited mobility and occurrence, tightly bound to colloidal particles like iron oxide, and when bounded with solid particles or dead remains of organisms it has restricted mobility and accessibility (Association 2017). In freshwater, in an acidic environment, Pb remains as a divalent cation (Pb^{2+}) while under basic medium it forms lead carbonate ($PbCO_3$) and hydroxide ($Pb(OH)_2$) (Association 2017). Pb^{2+} acts as a Ca^{2+}analog, and it is uptake by the gills of freshwater fish. It is more sensitive to the embryos and fries than that of the adults (Association 2017; Martin and Griswold

2009). In cyprinid fish (*Labeo rohita)*, it causes modification in the ultrastructure of gills (Brraich and Kaur 2015). In algae, Pb inhibits the enzyme needed for photosynthesis when its concentration exceeds 500 ppb (Jaishankar et al. 2014). Environmental Protection Agency (EPA) considers Pb as a carcinogen resulting in alteration of behavior, physiological processes, and adverse biochemical consequences within the human body (Association 2017). Fetuses and children below the age of six are more susceptible to its toxicity. It inhibits the production of hemoglobin; damage kidneys and reproductive systems, causing myocardial infarction; and also disrupts nervous coordination (Association 2017; Jaishankar et al. 2014; Martin and Griswold 2009). It can cross simply across the placenta leading to miscarriages, stillbirths, and neo-natal born with neurological damages (Jaishankar et al. 2014; Martin and Griswold 2009). Neurological damages result in lower IQ, hyperactivity, and poor attention span (Dhara et al. 2019). Prolong exposure results in illness that is acute or chronic. Acute exposure ends up in a loss of appetite, headache, hypertension, intestinal pain, kidney damage, tiredness, insomnia, arthritis, hallucinations, and vertigo. Chronic exposure implicates severe problems related to both mental and physiological damage (Tchounwou et al. 2012). It often shows mental retardation since birth, psychosis, autism, dyslexia, loss of body weight, muscular weakness, brain damage causing paralysis, nephropathy, and in adverse conditions cause death (Martin and Griswold 2009), (Tchounwou 2012). It usually happens within the employees that involve Pb production and processing industries. The priority list for the toxic effect of heavy metals are presented in (Table 10.2).

2.2.3 Arsenic

Arsenic (As) is the top priority global pollutant according to the ATSDR list (www.atsdr.cdc.gov/spl) existing as organic and inorganic form out of which inorganic form is the most toxic. Arsenite (As^{3+}) and arsenate (As^{5+}) are widely present inorganic forms. As^{3+} is the main reason for As toxicity in water as it is more toxic than arsenate and can disrupt enzyme functions (Goswami et al. 2011). Arsenic toxicity occurs due to various geological and anthropological processes including mining and ores processing (Ventura-Lima et al. 2011). Arsenic induces a variable degree of damages and toxic alterations in a riverine ecosystem that includes microbes, animals as well as aquatic plants. Alteration of toxicity in the bacteria is one of the common effects of As pollution due to surface water contamination (Goswami et al. 2015). The bacterium *Aeromonas hydrophila* isolated from fish shows acute to chronic toxicity due to exposure to As (Goswami et al. 2011; Ventura-Lima et al. 2011). The major alterations are inhibition of growth, changes in the hemolytic potential that results in the leakage of cytoplasm leading to cytoskeleton damage, alteration of pathogenicity, and induction of antibiotic resistance. Sodium As^{3+} can induce cytotoxicity in the fish cell line (Ventura-Lima et al. 2011). It competes with inorganic phosphate (PO_4^{3-}) and also binds to pyrophosphate moiety inflicting DNA damage (Goswami et al. 2011). Once exposed to As_2O, the Polychaeta *Laoenereis acuta* and *Clarias batrachus* show oxidative stress leading to the lowering of

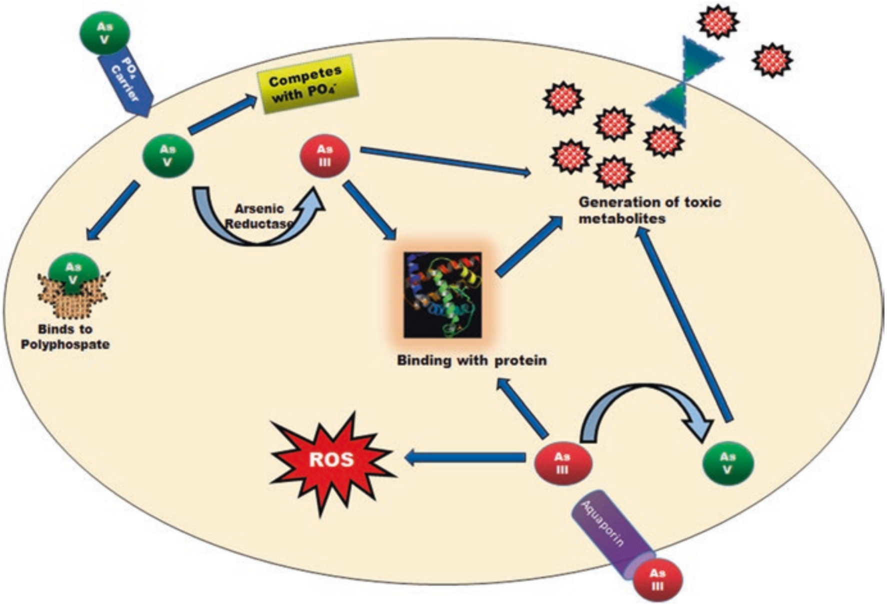

Fig. 10.3 Overview of the transformation of arsenic and toxic effects induced by the different forms of arsenic on the living organisms of the riverine ecosystem. Binding to physiologically important proteins, generation of toxic metabolites, and disruption of redox homeostasis are the key events of arsenic toxicity

antioxidative activities (Bhattacharya and Bhattacharya 2007; Ventura-Lima et al. 2011). As^{5+} rises the antioxidant responses within the gills of zebrafish *Danio rerio* (Cyprinidae) after 2 days of exposure (Seok et al. 2007). Goldfish (*Carassius auratus*) once exposed to As^{3+} leads to oxidative stress and inflection in the antioxidant system across the liver cells (Bagnyukova et al. 2007) and additionally in numerous tissues of common carp (*Cyprinus carpio*) on exposure to each of As derivatives (Wang et al. 2004). In certain fishes, As causes tissue-specific immune alteration and B cell T cell impairment (Goswami et al. 2015). As exposure via water is known to display both acute and chronic toxicity in humans and induces several health hazards (Jaishankar et al. 2014). Chronic As toxicity shows pigmentation and keratosis and is termed Arsenicosis (Jaishankar et al. 2014). Lower contamination of As causes nausea and vomiting, decreases the generation of erythrocytes and leukocytes, irregular heartbeat, pricking sensation across the limbs, and blood vessel damage (Tchounwou et al. 2012). Long-term exposure results in the formation of sores in the skin, carcinomas, nervous disabilities, heart problems, peripheral vascular dysfunction, high blood pressure, and diabetes mellitus (Jaishankar et al. 2014). Carcinogenic outcomes leading to cancers of lungs, liver, bladder, and skin are also included as the principal hazardous effects of the As toxicity (Tchounwou et al. 2012), (Goswami et al. 2015; Jaishankar et al. 2014). An overview of the arsenic-mediated toxicity in the organisms of the riverine ecosystem is given in Fig. 10.3.

2.2.4 Mercury

Mercury(Hg) is another hazardous heavy metal known for causing toxic alteration whenever it gets a chance to interact with the biologicals systems, of which methyl mercury is the highest toxic form. The methylation of inorganic Hg into its organic form is facilitated via lower pH and higher dissolved organic carbon resulting in accumulation at the water-sediment interface (Yuan et al. 2017; Wright and Welbourn 2002). Sources of Hg into the riverine ecosystem are from the discharge of various industries like paper and pulp; preservatives, chlorine, and caustic soda production industry; and agriculture waste and pharmaceutical industries (Jaishankar et al. 2014). Methyl-mercury crosses biological membranes due to their lipophilic nature and perseveres within the fatty tissue resulting in bio-concentration and biomagnification (Schneider et al. 2009; Hsu-Kim et al. 2013). Each level of the food chain increases the concentration of its prey resulting in thousands or millions of times greater in the top than in water or sediments (Hsu-Kim et al. 2013). Acrodynia or pink disease is a disease that occurs due to Hg poisoning. Bioconcentration and biomagnification of Hg depending on the age, size, species of fish, and duration of exposure (Jaishankar et al. 2014; Tchounwou et al. 2012). In the Red-headed river turtle (*Podocnemis erythrocephala*), bioaccumulation chiefly occurs in soft tissues and the liver (Schneider et al. 2009).

In humans, inorganic Hg leads to miscarriage, inborn malformation, and gastrointestinal disorders causing corrosion to the esophagus and hematochezia (Jaishankar et al. 2014). Organic forms of Hg, which include monomethyl and dimethyl-mercury lead to poisoning inflicting erethism, depletion of memories, tremors, tiredness, headache, hair loss, and so on (Jaishankar et al. 2014). Minamata disease is caused by a higher level of Hg toxicity. Hg is additionally working as a neurotoxin leading to structural damage of the brain causing nervousness, memory difficulties, irritability, vision and hearing impairment, and inhibiting neurotransmission even at a lower concentration of 10 ppm (Wright and Welbourn 2002). The developing brain of children and fetuses is mainly at a higher risk. Hg can even cross the placenta and cause Minamata disease in newborns (Jaishankar et al. 2014; Tchounwou et al. 2012; Wright and Welbourn 2002). Contamination with a higher concentration of metallic Hg vapors for even shorter periods results in damage to the respiratory system including lungs, vomiting, diarrhea, nausea, skin lesions, tachycardia, and increased blood pressure (Jaishankar et al. 2014; Tchounwou et al. 2012).

2.2.5 Nickel

Nickel (Ni) is usually occurred in soils and volcanic rocks and is employed in many industrial utilizations consisting of electroplating, automobile manufacturing, aircraft fragments, cosmetics, stainless steel, batteries, coins, and spark plugs and are extensively used for the generation of nickel–cadmium batteries (Factor and de Chavez 2012). It discharges within the aquatic ecosystem through the natural phenomenon of weathering of rocks and soils causing leaching of minerals and discharge of waste from industries. The water-soluble salts of Ni are the major contaminant in riverine systems (Brix et al. 2017). Although it is essential within the synthesis of red blood cells; however, higher doses cause toxicity. Trace amounts of Ni have no ill effect in biological cells; however, prolonged exposure to higher doses may damage cells, causing decreased bodyweight due to a reduction in cell growth, impairment of liver and heart, damage to the nervous system, and also leads to cancer (Verma 2012). In the fish, *Hemichromis fasciatus* from the Ogba river, Ni bioaccumulation was significantly variable $p < 0.05$ and is season-specific (Madoni 2000). Ni toxicity in the riverine ecosystem mainly disrupts Ca^{2+}, Mg^{2+}, and $Fe^{2+/3+}$ homeostasis that induces production of (ROS) and oxidative damage (Brix et al. 2017; Factor and de Chavez 2012). Consequences result in immunological dysfunction, various allergic reactions in the respiratory epithelia, and also apoptosis (Factor and de Chavez 2012). In humans, it mainly affects the liver and heart causing a reduction in cell growth, damage of the antioxidant system, and generation of various type II enzymes (Martin and Griswold 2009).

2.2.6 Barium

Barium (Ba) is a divalent alkaline heavy metal known to be a muscle poison. Ba and insoluble Ba salts, like Ba sulfate, do generally not act as a pollutant, but water-soluble forms including Ba chloride, Ba carbonate, Ba nitrate, and Ba acetate can form a dimer with organic compounds showing greater toxicity (Neff 2008). The primary source of Ba contamination occurs by the usage as a weighting agent in the drilling process to counter the pressure and prevention from blowout (Neff 2008). In freshwater alga (*Chlorella sp*.) and water flea (*Ceriodaphnia dubia*), Ba accumulates in the muscle and soft tissue causing toxicity. It results in biomagnification in the higher organisms across the food chain (Golding et al. 2018). In humans, it affects the skeletal, smooth, and even cardiac muscle causing myocardial excitability resulting in hypokalemia; affects the respiratory system causing secondary paralysis; and malignant arrhythmia leading to severe illness (Tao et al. 2016). In acute toxicity, the blood concentration elevated within 12 hours of exposure showing poisoning after 24 hours, resulting in atrioventricular blockage, ventricular tachycardia, prolonged PR interval, ST-segment depression across the U waves, and inversion of T-wave (Tao et al. 2016). Other symptoms include gastrointestinal burning, nausea, vomiting, vertigo, and weakness.

2.2.7 Chromium

Chromium (Cr) is an important nutrient and is necessary for an adequate amount of metabolism of carbohydrates. However, higher concentrations lead to toxicity of which a significant bioactive toxic form is the Cr^{+6} hexavalent ions. Although Cr does not undergo biomagnification, it shows bioconcentration (Velma and Tchounwou 2010). Hexavalent Cr at even low concentration can induce sublethal

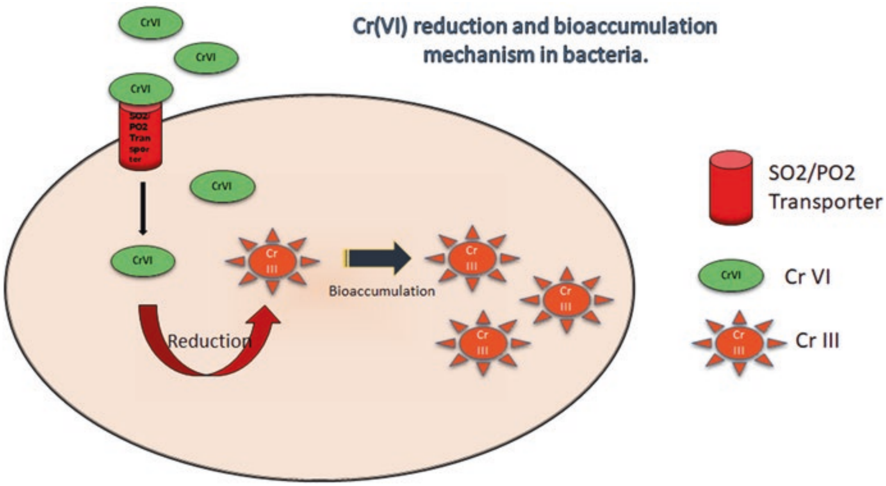

Fig. 10.4 Mechanism of bioaccumulation of Chromium (Cr(VI)) in bacteria. Metallic Chromium is entered into the bacterial cells by the action of SO2/PO2 transporter and converted into Cr(III) which is accumulated in the cell

toxicity in aquatic plants and animals. It inhibits growth in Chinook salmon at 16 ppb and various algae at 62 ppb (Velma and Tchounwou 2010). In humans, it causes severe health issues causing sores in the mouth, bleeding of the nose, renal dysfunction showing nephritis, and can be fetal at 10 mg/kg of body weight (Velma and Tchounwou 2010). It lowers white blood cell counts causing depressed immune defense systems. Chronic toxicity results in miscarriage and also neonatal shows deformed spines and bone (Factor and de Chavez 2012; Wright and Welbourn 2002). One of the major attributes of Cr toxicity is the bioaccumulation of this metal in living organisms, especially in microbes, and this is considered as the principal reason behind the pathological outcomes of Cr toxicity. The mechanism of Cr bioaccumulation in bacteria is presented in Fig. 10.4.

Different types of heavy metals, their target organisms, and their damaging effects on the organisms of the riverine and extra-riverine ecosystem have been presented in Table 10.3.

The mechanistic insights of induction of the toxicity in the aquatic animals and microbes by different heavy metals are presented in Fig. 10.5.

3 Impact of Riverine Pollution on the Riverine and Extra-Riverine Ecosystem

In the earlier section, we have presented an overview of the different types of organic and inorganic pollutants that contaminate and hamper the riverine ecosystems. We have discussed the molecular mechanism of the health hazards caused by the

Table 10.3 Effects of heavy metals on the riverine ecosystem

Heavy metals	Organisms effected	Mode of action and extent of damage	References
Cadmium (cd)	*Photobacterium phosphoreum*, *Daphnia magna*, and *Carassius auratus*	Induction of oxidative damage	Qu et al. (2013)
	*Caelatura companyoi and Cleopatra bulimoide*s	Bioaccumulation and toxicity	Moloukhia and Sleem (2011)
	Freshwater fish (*Oreochromis niloticus*)	Dysregulation of glutathione metabolism	Eroglu et al. (2015)
	Fishes	Bioaccumulation in liver and kidney, reproductive dysfunction; immune dysfunction through induction of apoptosis in macrophages and T cells	Kitana and Callard (2008)
	Humans	Itai-Itai disease, renal dysfunction, cadmium pneumonitis, bone defects, and myocardial dysfunctions	Tchounwou et al. (2012)
Lead (Pb)	Cyprinid fish (*Labeo rohita)*	Ultrastructural changes in the gills.	Brraich and Kaur (2015)
	Algae	Inhibition of photosynthesis	
	Humans	Intellectual debility, intestinal pain, constipations, headaches, irritability, loss of memory, infertility, and tingling in the limbs	Dhara et al. (2019), Tchounwou et al. (2012)

(continued)

Table 10.3 (continued)

Heavy metals	Organisms effected	Mode of action and extent of damage	References
Arsenic (as)	Juvenile rockfish, *Sebastes schlegelii*	Induction of oxidative stress	Kim and Kang (2015)
	Bacterium (*Aeromonus hydrophilla*, *Acenetobactor junii*, *Acenetobactor baumanni*)	Leakage of cytoplasm, alteration of pathogenicity, and induction of antibiotic resistance	Goswami et al. (2015)
	Goldfish (*Carassius auratus)*	Induction of oxidative stress and modulation of the antioxidant system in the liver	Bagnyukova et al. (2007)
	Polychaete (*Laoenereis acuta)* and fish (*Clarias batrachus)* zebrafish *Danio rerio* (Cyprinidae)	Induction of oxidative damage and depletion of antioxidant enzyme activities	Bhattacharya and Bhattacharya (2007), Ventura-Lima et al. (2011)
	Fish	Tissue-specific immune alteration and impairment of B- and T-cell activity	Seok et al. (2007), Wang et al. (2004)
	Humans	Arsenicosis, pigmentation and keratosis, skin lacerations, carcinomas, neurological difficulties, pulmonary disease, hypertension, and cardiac illness, and diabetes mellitus	Martin and Griswold (2009)
Mercury (hg)	Freshwater crayfish; *Channa punctatus*	High toxicity leads to death	Del Ramo et al. (1987)
	Red-headed river turtle (*Podocnemis erythrocephala*)	Bioaccumulation	Schneider et al. (2009)
	Fishes	Bioconcentration and biomagnification	Wright and Welbourn (2002)
	Humans	Minamata disease and neurotoxin	Tchounwou et al. (2012)

Nickel (Ni)	Ciliated protozoa (*Spirostomum teres* and *Paramecium bursaria*)	Acute toxicity and bioindicator	Madoni (2000)
	Juvenile fathead minnows (*Pimephales promelas*)	Bioaccumulation and toxicity	Lapointe and Couture (2009)
	Fishes	Toxicity and sensitivity	Factor and de Chavez, (2012)
	Humans	Results in the damage in the liver and heart and reduction of cell growth	Brix et al. (2017)
Barium (Ba)	Freshwater alga (*Chlorella sp.*) and water flea (*Ceriodaphnia dubia*)	Bioaccumulation and toxicity	Golding et al. (2018)
	Humans	Muscle poison, myocardial excitability, and hypokalemia	Tao et al. (2016)
Chromium (Cr)	Chinook salmon and various algae.	Inhibition of growth	Velma and Tchounwou (2010)
	Carassius auratus; Channa punctatus, Clarias gariepinus	Histopathological alterations in gill, liver, and kidney	Velma and Tchounwou (2010)
	Humans	Sores in mouth, nasal bleeding, renal problems, and decreased white blood cell counts	Velma and Tchounwou (2010), Wright and Welbourn (2002)

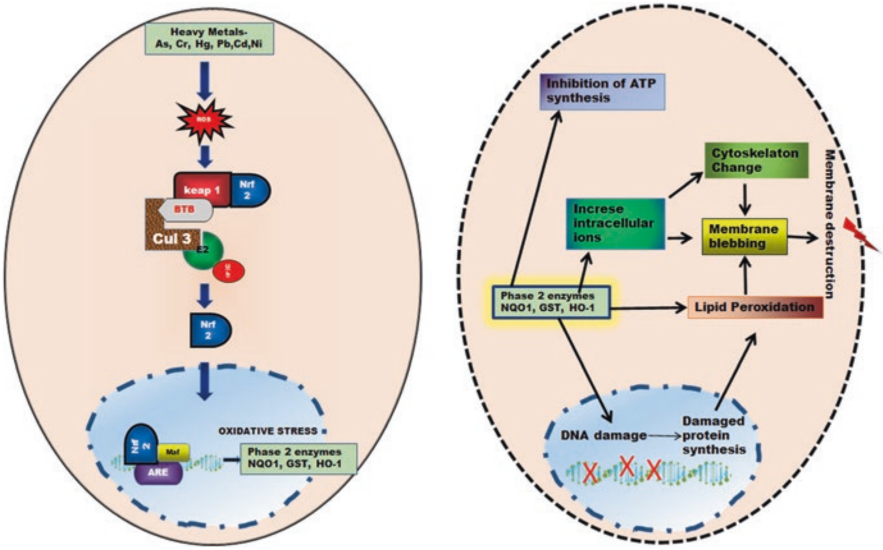

Fig. 10.5 Generalized model on the mechanistic insights of cytotoxicity induced by different heavy metals on the different living organisms of riverine ecosystems. Most of the heavy metals trigger disruption of the antioxidant system, thereby causing a breakdown of the cellular redox homeostasis. Toxic metals interact with the Nrf2 signaling pathway and result in overexpression of phase 2 detoxifications proteins/enzymes. All these events cumulatively increase intracellular ion concentration, damages of membrane and cytoskeleton, oxidative stress, and DNA damage leading to cell death

contaminants. Therefore, it is extremely important to study the impact of the aforesaid contaminants on the ecological attributes of riverine and extra-riverine ecosystems. These have been discussed in this section.

3.1 Behavioral Abnormalities Caused by Freshwater Pollution

Many components of freshwater pollution alter the behavior of the resultant animal, thus affecting the ecosystem, food chain, and eventually the survivorship of the group. Ecosystem stability can disrupt significantly as the food chain and biodiversity is hugely affected by the over-active, aggressive, and altered foraging behavior of its components or by slow and low activity or predator response or decreased feeding leading the animal to predation pressure or reducing its reproductive success (Gherardi 2002; Manning et al. 2012; Schmitz 2007). Furthermore, behavior can be very sensitive to environmental stresses and also connect physiological processes to important ecological functions and components like food chains, trophic balance, reproductive success, survivorship, mate selection, and sociability. The environmental stressors that affect the behavior of freshwater organisms include pharmaceuticals and personal care products (PPCPs), herbicides, insecticides, heavy metals, micropollutants, and so on.

Pharmaceuticals and personal care products (PPCPs) are unique among themselves due to their property of inducing effect at minimal doses in their primary target, that is, humans. As such, the amount that is run-off to the natural water bodies is significant and can show its effects on non-target organisms. The most common PPCPs include antibiotics, hypertension drugs, antidepressants, or psychiatric drugs like selective serotonin reuptake inhibitors (SSRIs), anxiolytic drugs, and so on, all of which impact the natural systems of different non-target organisms to different magnitude altering their physiology and behavior, making them susceptible to ecological stresses. SSRIs are especially important in modulating receivers' behavior to a different extent. Fundamental behavioral traits like boldness, feeding behavior, activity, social responses, fight, or flight responses all can shift from standard in response to PPCPs' presence and which in turn change ecosystem functioning of the freshwater system in concern (Manning et al. 2012; Schmitz 2007). Ample research data are available to support the adverse effects of PPCP pollution in water. Very low environmentally available concentrations of pharmaceutically active compounds (PhAC) can significantly affect the behavior of aquatic invertebrates (Grabic et al. 2012; Grabicova et al. 2015).

Antidepressants or psychiatric drugs are reported to have extensive effects on the rate of feeding as well as general activities of organisms (Backhaus 2014; Jonsson et al. 2014). Crayfishes in the presence of Tramadol and Citalopram displayed lower velocity and eventually traveled shorter distances than unexposed ones. The exposed crayfishes showed higher shyness and added to this the reduced mobility caused by spending more time inside the shelter (Ebele et al. 2017). Venlafaxine and Citalopram also showed to induce bivalves (*Leptoxiscarinata* and *Stagnicola* (*Lymnaea*) *elodes*) for behavioral change like foot detachment from the substrate (Fong and Hoy 2012). Several fish species like that of three-spined stickleback show effects such as increased locomotor activity due to the presence of Citalopram at very low concentrations (Kellner et al. 2016).

Fluoxetine and other SSRIs that target the serotonin signaling pathway are reported to have diverse behavioral alteration effects on aquatic organisms (Hazelton et al. 2014). The behavior of freshwater mussels alters substantially in response to extended exposure to fluoxetine. Fluoxetine-treated mussels showed a significant increase in distance covered. Time for burrowing decreased and an increased daytime movement was also observed. Lure display behavior was also affected positively in treated mussels. This along with altered movement patterns might affect their prey capturing capabilities. Furthermore, as the movement during daylight increases, the mussels get more susceptible to predation affecting the local population size considerably (Hazelton et al. 2014). In Crustacea, exposure to fluoxetine significantly decreased the general activity of *Gammarus pulex* (Crustacea, Amphipoda) (De Lange et al. 2006). Fluoxetine, as it regulates serotonin level, subsequently influences numerous behavioral components such as aggression, fear, locomotion, and feeding in Arabian killifish (*Aphanius dispar*) and *Betta splendens*. Fluoxetine is reported to lessen aggression or chasing behavior, average swimming velocity as well as swimming speed while responding to predator alarm signals (Barry 2013; Kohlert et al. 2012). Fluoxetine also has a detrimental effect on the swimming behavior of mosquitofish *Gambusia holbrooki*. Various behavioral effects in fish, for example, weakened mating behavior of *Pimephales promelas* (fathead minnow) (Weinberger II and Klaper 2014), decreased territoriality in coral reef fish, Siamese fighting fish, Arabian killifish and round goby *Neogobius melanostomus* (Barry 2013; Dzieweczynski and Hebert 2012; Kania et al. 2012; Kohlert et al. 2012; McCallum et al. 2017; Perreault et al. 2003; Weinberger II and Klaper 2014) or increased in toadfish (McDonald et al. 2011); decreased brood defense in *Betta splendens* (Forsatkar et al. 2014; Greaney et al. 2015); lowered anxiety in zebrafish or medaka *Oryzias latipes* (Ansai et al. 2016); decreased stress response in zebrafish (de Abreu et al. 2014); reduced locomotor activity in *Betta splendens*, sheepshead minnow *Cyprinodon variegatus*, or Arabian killifish (Kohlert et al. 2012; Weinberger II and Klaper 2014); and lowered predator avoidance in fathead minnows, guppies, or *Gambusia holbrooki* (Martin et al. 2017; Painter et al. 2009; Pelli and Connaughton 2015) can be seen affected by SSRIs like Fluoxentine. A significant alteration of standard behavior following antihistamine exposure was seen. After antihistamine drugs like Hydroxyzine or Fexofenadine alter the behavior of damselfly larvae significantly leading to lower activity and reduced fleeing response leading to increased boldness (Jonsson et al. 2014).

Pharmaceuticals like ibuprofen (a nonsteroidal anti-inflammatory (NSAID) drug) and furosemide (diuretic drug) are known to significantly increase larval activity in *Diamesa* (Diptera, Chironomidae), whereas the antibiotic trimethoprim decreased average speed and distance moved by the larvae (Villa et al. 2018). Activity in *Gammarus pulex* (Crustacea, Amphipoda) also reduced significantly due to exposure to ibuprofen as well as to anti-epileptic drug carbamazepine and a surfactant CTAB resulted (De Lange et al. 2006). Carbamazepine also caused inhibition of siphoning behavior in the Asian clam *Corbicula fluminea* (Chen et al. 2014). Environmentally significant concentrations of the anxiolytic drug Oxazepam also modulate the feeding rate in European perch (*Perca fluviatilis*). Behavioral changes include increased activity, as well as reduced social behavior, and increased bolder, which eventually resulted in a greater feeding rate (Brodin et al. 2013). Zhang et al. 2019 have reported that *Xenopus laevis* tadpoles when exposed to a pyrethroid insecticide, cis-bifenthrin, an initial decline in the percentage of individuals breathing surface air is noted but it increased after a few days (Zhang et al. 2019). The activity decreased in all the treatments. The tadpoles lacked proper equilibrium posture which resulted in lateral or dorsal-lying or swirling behavior. The feeding also decreased during exposure. The inverse behavioral shift from initial to long-term exposure may be due to the adjustment by the tadpoles to the toxic environment (Zhang et al. 2019). In response to exposure to an organophosphate pesticide, Chlorpyrifos, Japanese medaka (*Oryzias latipes*) showed hyperactivity and elevated anxiety. Unusual startle responses to a sudden stimulation were also noted. The swimming distance was also affected due to exposure (Qiu et al. 2017). On the other side, behavioral responses of freshwater carp, *Cyprinus carpio,* in presence of Chlorpyrifos, exhibited irregular, erratic, and darting swimming patterns. The fishes were also hyper-excited, with equilibrium lost and sank to the bottom (Halappa and David 2009). Low concentration of an anthranilic diamide insecticide, chlorantraniliprole, significantly decreases behavioral attributes like predation and feeding, locomotor activities, and total distance moved, in the freshwater planarian *Dugesia subtentaculata* (Rodrigues et al. 2016). Asian swamp eel (*Monopterus albus*), due to their muddy habitat, are prone to rice field insecticides like endosulfan. The toxicity alters the behavior of the eel as the swimming pattern shifts to more irregular with restlessness, tremors, and erratic movements of the body, increased lethargy, and with quick tiredness leading to bottom sinking and no opercular movements (Hii et al. 2007). Acute toxicity of dimethoate, an organophosphate pesticide, caused enhanced opercular movement and abnormal unenergetic inconsistent swimming in freshwater catfish *Heteropneustes fossilis*. Muscular spasm and difficulty in floating were also common in the stressed fishes (Srivastava et al. 2010). Another group of major toxicants are heavy metals that alter the standard behavior of organisms significantly. Freshwater worm, *Branchiura sowerbyi*, in response to Lead (Pb) toxicity showed a significant decrease in normal clumping and rapid twisting behaviors with the increasing concentration (Dhara et al. 2019). It is reported that exposure to Lead (Pb) and Copper (Cu) caused increased ventilation and decreased locomotion in the crustacean *Gammarus pulex*. Cd-exposed *Gammarus pulex* also demonstrated significantly reduced behavioral activities including feeding rate, locomotor activities, and ventilation (Felten et al. 2008). In a study with two freshwater prawn species, *Macrobrachium lamarrei* and *Macrobrachium dayanum,* it was observed that increased concentration of Nickel in the medium elevated aggression and affected balance in both prawn species (Verma 2012). Another study with sub-lethal doses of Cd and Zinc (Zn) showed that with increasing concentrations the feeding rate of two freshwater crustaceans, *Atyaephyra desmarestii* (Decapoda) and *Echinogammarus meridionalis* (Amphipoda), significantly reduced (Pestana et al. 2007). The effects of Cd and manganese (Mn) pollution on the snail *Biomphalaria alexandrina* shows a significant decrease in attachment, locomotion, and feeding behaviors (Habib et al. 2016). With increasing concentration of Cd, different behaviors like the clumping tendency, crawling activity, and reflex of the exposed freshwater snail *Lymnaea acuminate* (Dhara et al. 2019). Behaviors like movement, the opening of valves or foot and siphon extension in freshwater bivalves *Lamellidens jenkinsianus obesa* and *Parreysia (Parreysia) corrugate* were noticeably impacted if Pb is present in the medium (Brahma and Gupta 2020). Fishes are one group that are extremely affected by metal pollution, vastly studied, and are of serious concern not only ecologically but also economically. As a large mass of people globally source significant amount of their protein requirement from fishes, toxicity in fishes may lead to a health hazard in humans directly (Singh et al. 2018). Mercury in an aquatic environment is known to cause erratic swimming, loss of balance, abnormal posture, loss of equilibrium, sluggish un-coordinated movement, slower swimming velocity, increased surface activity, increased opercular movement, convulsions and excessive mucus secretion over the body surface, and low antipredator activities in different fishes (Siddiqui and Arifa 2011; Webber and Haines 2003). Acute lethal and chronic sublethal treatments with Hg, Cd, and Pb on the fish *Tinca tinca* showed various behavioral shifts. Acute doses of Hg showed increased breathing rate and surfacing activity initially, irregular, erratic and jerking movement, loss of equilibrium, and unconsciousness on later stages. Acute Cd and lead exposure caused hyperactivity, restlessness, and surfacing behavior initially, with sinking of fish was common later. Chronic sublethal treatment with Hg resulted in restlessness with fast fin movement and floating in a vertical posture with mouth upward near the surface exposure (Shah and Altindag 2004). With time lethargy came in with decreased feeding activity. Cd caused erratic restless movements, increased breathing, initially, but vertical floating with minimum movement at later stage like that of Pb exposure (Shah and Altindag 2004). Acute toxicity of Lead Nitrate metal salt resulted in hyperactivity, loss of balance, disoriented swimming patterns, convulsions, and increased mucus secretion in the Milkfish (*Chanos chanos*). High concentrations of Pb also caused reduced movement and difficulty in breathing (Hesni et al. 2011). Rahman et al. 2016, found similar behavioral alterations like erratic swimming, lack of balance, and initial hyperactivity, but lethargy at later stage, surfacing activity, and decrease in feeding due to Zn toxicity in freshwater carp, *Labeo rohita* (Rahman et al. 2016). Freshwater fish, *Percocypris pingi*, on exposure to Hg and Cd initially showd hyperactivity which ceased with time (Yuan et al. 2017). These examples show widespread effects on the behaviors of almost every group of aquatic animals by exposure to some very common pollutants.

3.2 Reproductive Abnormalities Caused by Freshwater Pollution

Reproduction is a very important tool in an ecosystem. Numbers are important in the case of maintaining the stability of the system, and the ecosystem is no different. Different components of the ecosystem, be it food chains and webs, diversity, complexity, sustainable functioning, population survivability, migration, or extinction. It is important to maintain the balance of population sizes across different levels of the food pyramid so that the normal function of an ecosystem can sustain. Among the three most important components that govern population size are natality, survivability, and mortality; reproduction controls the basic one: natality. Pollution impacts all three components of population growth altering the reproductive behavior or success of the organism, making it vulnerable and reducing its survivability, and thus increasing mortality production (Flaherty and Dodson 2005). Different freshwater pollutants like pharmaceuticals, personal care products, insecticides or heavy metals, and so on impact the reproductive properties of different groups of animals.

A metabolite of an anticholesteremic drug Clofibrate, Clofibric acid, increases the production ratio of male offspring in a *Daphnia* population (Flaherty and Dodson 2005). Serotonin is reported to affect the invertebrate sex hormones:

ecdysteroids, ecdysone, and juvenile hormones, and also modulate oogenesis. Thus, the very commonly found SSRI, Fluoxetine, may act as endocrine disruptors in invertebrate systems increasing fecundity, as seen in *Daphnia*. The antimicrobial drug Triclosan also induced a change in sex ratio by more male production (Flaherty and Dodson 2005). The SSRIs, fluvoxamine, fluoxetine, and paroxetine, are potent inducers of spawning in any bivalve at low concentrations, and fluvoxamine being the strongest among all as seen in Zebra Mussel (*Dreissena polymorpha*). An increased number of egg masses in *Physa acuta* is reported at lower concentrations, whereas, at a higher level, a reduced reproduction was observed (Sánchez-Argüello et al. 2009). Fluoxetine induced parturition in the freshwater mussel *Elliptio complanata* also, although most of the larvae could not survive. In males, the pharmaceutical induced spermatozeugmata release (Bringolf et al. 2010). Very low concentrations of a more potent metabolite, norfluoxetine, are known to induce untimely spawning and parturition in zebra mussels and fingernail clams, respectively (Fong and Molnar 2008). The NSAID diclofenac overall decreases the reproduction in the crustaceans, *Daphnia magna*, and *Moina macrocopa* and also affects the hatching of eggs of the fish, *Oryzias latipes* (Lee et al. 2011). In another study, with increasing concentration of another NSAID, Ibuprofen, frequency of spawning decreased, although the egg count per spawning increased in Japanese medaka (*Oryzias latipes*) which indicates reproductive damage and compensatory response. Increased concentration of the NSAID also delayed the hatching of eggs, making them prone to predation (Han et al. 2010). The herbicide Atrazine also increased the male sex ratio in a population of *Daphnia pulicaria* (Flaherty and Dodson 2005). Atrazine also reported effects in gonadal morphology, sex hormone concentration, and gonadal function like spermatogenesis in cases of fish and amphibians (Rohr and McCoy 2010). During sexual differentiation, exposure to atrazine caused a decreased volume of the testis, and a decrease in several primary spermatogonial cell nests as well as nutrition-providing nurse cells in *Xenopus laevis* tadpoles. Testicular resorption, as well as underdeveloped testis, was observed. Atrazine is also reported to decrease testosterone levels by converting androgens to estrogens and thus promoting demasculinization in males resulting in hermaphrodites. Similar results of male demasculinization and hermaphrodite production were also reported in leopard frogs (*Rana pipiens*) (Hayes et al. 2002; Tavera-Mendoza et al. 2002). Juvenile hormone analog (JHA) insecticides like pyriproxyfen shifted the sex ratio toward a more male population in *Daphnia magna*. This type of pesticide also impacts the life expectancy and fecundity of the water flea *Moina macrocopa* (Hu et al. 2020). Pyrethroid insecticide, esfenvalerate, markedly alters reproductive properties like promptly separated reproducing pairs, immediate parturition, and notable hindrances during future pair-formation, and reproduction of the freshwater amphipod, *Gammarus pulex* (Cold and Forbes 2004). Deltamethrin, on the other hand, affects reproduction in daphnids as it decreases multiple life-table traits, namely, longevity, total molt number, rate of population growth (r), number of broods, and fecundity (Toumi et al. 2013). Other pyrethroid insecticides, cypermethrin, and alphamethrin, hugely affect the reproductive properties of the freshwater snail, *Lymnaea acuminate*. Although the numbers of egg masses and eggs are increased, the ultimate survivability of the snails decreases significantly impacting the population size. With an increased dose of deltamethrin, reduced hatching success and increased larvae death were commonly noted in common carp, *Cyprinus carpio* (Köprücü and Aydın 2004).

Antimicrobial additive Triclosan (TCS) has wide-field effects on several organisms ranging from micro-crustaceans to amphibians and fishes. Most of the effects are reproductive or related to changing the population dynamics of the concerned organism. A significant reduction in reproduction (number of neonates produced) is noted in *D. magna* in response to Triclosan (Orvos et al. 2002). In the copepod *Tigriopus japonicus*, reduced fecundity can be induced by TCS (Park et al. 2017). The life table parameters in the case of two Rotifers (*Plationus patulus* and *Brachionus havanaensis*) showed significant alterations in response to TCS exposure. Age-specific survivorship, Fecundity, gross reproductive rate (GRR), or net reproductive rates (NRR) all declined in response to TCS. However, the mean generation time was shorter in the exposed individuals (González-Pérez et al. 2018). In case of fishes, like medaka (*Oryzias latipes*), triclosan exposure delayed the hatching time as well as reduced the hatchability of eggs. The gonadosomatic index (GSI) also increased after exposure. Vitellogenin (VTG) concentration in males also increased, showing its weak estrogenic potential (Ishibashi et al. 2004). Another study on western mosquitofish, *Gambusia affinis*, also showed the endocrine-disrupting potential of Triclosan. Following exposure, a significant rise in female-specific VTG gene expression in male fishes as well as a significant reduction in sperm count was noted (Raut and Angus 2010). TCS also caused a delayed metamorphosis in larval zebrafish, and after metamorphosis, reduced reproductive fitness persists (Stenzel et al. 2019). A study with the frog, *Pelophylax nigromaculatus*, reported that exposure to TCS caused gonadal deformities like very few oocytes and a lot of somatic cells. TCS also seemed to induce early metamorphosis (Chen et al. 2018). Embryos of the Chinese toad, *Bufo gargarizans*, when exposed to TCS showed delayed development as well as teratogenic effects such as hyperplasia, abdominal edema, and axial flexures (Chai et al. 2016). Another group of antimicrobial agents, parabens (e.g., isopropyl-, propyl-, methyl-, ethyl-, isobutyl-, butyl-, benzyl paraben, etc.) are very commonly used preservatives in food, cosmetics, toiletries, and pharmaceuticals. This group has several disruptive reproductive effects like detrimental effects on reproduction of *D. magna*, slight estrogenic effects, and VTG synthesis in fishes (Brausch and Rand 2011; Darbre et al. 2003; Inui et al. 2003).

Another growing concern in the aquatic system is the ever-increasing contamination of UV filtering materials used in sunscreen products and cosmetics that can be of both inorganic and organic origin. UV filters make their way into aquatic systems via run-offs from wastewater treatment facilities or direct infiltration from recreational activities. UV filters are potent endocrine disruptors as proven by several studies both in vitro and in vivo. The effect of the UV-filter on 2-ethyl-hexyl-4-trim ethoxycinnamate (EHMC) has been found to cause a deleterious effect on two freshwater snails, *Potamopyrgus antipodarum* and *Melanoides tuberculate*, as in both cases the embryo production reduced significantly with exposure (Kaiser et al.

2012). 3-benzylidene-camphor (3-BC) and 3-(40-methylbenzylidene)-camphor (4-MBC) also have been reported for affecting the reproduction in aquatic oligochaete *Lumbriculus variegatus*, as both UV filters decreased reproduction (Kaiser et al. 2012). Significant dose-dependent overexpression of the ecdysone receptor gene (*EcR*) was noted in *Chironomus riparius* following exposure to EHMC, 4-MBC, and octyldimethyl-p-aminobenzoate (OD-PABA) (Ozáez et al. 2013). UV screens are known for their estrogenic activity. As many as 10 compounds showed estrogenicity in fish in in vitro studies. In studies using fish models (*Pimephales promelas*, *Oncorhynchus mykiss*), it was found that UV filters not only affect the estrogenicity but also reduces the fecundity and reproductive capacity of the fishes. Compounds like 3-Benzylidene camphor (3 BC), benzophenone-1 (BP1), benzophenone-2 (BP2), and octocrylene (OC) induce VTG synthesis in dose-dependant manner in *O. mykiss* and *O. latipes*. (Coronado et al. 2008; Holbech et al. 2002; Kunz et al. 2006). Benzophenone-3 (BP-3) acts as a potent endocrine-disrupting agent, increasing plasma concentration of testosterone in male *O. latipes*, down-regulating steroidogenic genes in gonads, and a daily average rate of egg production per female also decreased (Kim et al. 2014).

Another group of major aquatic toxicants is heavy metals that affect several freshwater organisms, including daphnids, snails, fishes, amphibians, and so on. Although the reproductive effects of metals are somewhat understudied, it is not unknown. Reproductive output, that is, fertile eggs per individual decreased significantly at higher concentrations. In another study, each one of As, Al, Cr, and Ni caused a reduction in hatchability and increased incubation period of eggs of exposed freshwater Snail, *Radix quadrasi*. Moreover, the shell thinning in embryos, as well as decreased embryo survivability, was noticed (Factor and de Chavez 2012). Gomot (1998) described that Cd^{2+}in different doses caused several reproductive and developmental impairments in the freshwater snail *Lymnaea stagnalis*. Dose as high as 400 µg/L caused halted egg production and reduced hatching up to 0.4% at the concentration of 200 µg/L (Gomot 1998). Egg development was blocked at the first cleavage stage by the highest concentration of Cd^{2+} with lower concentrations affecting different other developmental stages of the eggs including gastrula, veliger, or pre-hatching. Hatching delays of about 5 to 15 days than the controls were also reported. When subjected to different doses of Cd^{2+}, the sperms of the freshwater crab *Sinopotamon henanense* encountered oxidative damages as reactive oxygen species (ROS) formed. Treating the crabs for 7 days with 116 mg/L Cd^{2+} caused the sperm cell membrane to nearly disintegrate, heavily condensed and irregular chromatin fibers in the nucleus, and major damage to the acrosome were also reported (Ma et al. 2013). It is also found that exposure to heavy metals (Hg, Cd, Pb, and As) can decrease the sex steroid hormones, estradiol in females and progesterone, and testosterone in male *C. carpio* and *Capoeta* sp. fishes (Ebrahimi and Taherianfard 2011). Sujata (2015) reported localization of Cd in the reproductive organs causes detrimental biochemical alterations and thus affecting the reproductive capacity in freshwater fish, *Channa punctatus* (Sujata 2015). Gonadal development in response to the presence of Cd was severely affected in two freshwater turtles (*Trachemys scripta* and *Chrysemys picta*) embryos. The total number of germ cells reduced to below half in the treated (1 μg/g) embryos than in controls which may reduce the proliferation migration of germ cells. Also, oocyte apoptosis was significantly increased ultimately reducing follicle number in adult turtles (Kitana and Callard 2008). Thus, the common pollutants in the freshwater system can alter the reproductive behavior of several groups of aquatic organisms, changing the population dynamics upside down. These results from different groups suggest how the pollution in freshwater ecosystems impacts one of the fundamental components that include the community structure and complexity. As a more complex community leads to a more stable ecosystem, shifting population dynamics can ultimately disintegrate the stability of the system, driving local or global extinction of many of its unit lives.

3.3 Effects on Bioaccumulation, Biomagnification, and Food Web

Chemical concentration in the environment and that in an organism is important to judge the organisms' responses to the situation as well as the toxicity of the chemicals. This correlation between the organism and the chemical concentration in its environment is usually measured using several terms like Bioaccumulation, Bioconcentration, Biomagnification, Trophic transfer, and so on. A simple way of measurement has been developed in terms of factors (e.g., Bioaccumulation factors, bioconcentration factors, and biomagnification factors) which gives the approximate of how a chemical is transferred from environment to the residents of that environment (Hoffman et al. 2002; Mackay et al. 2018).

3.3.1 Bioaccumulation, Bioconcentration, Bioaccumulation Factor, and Bioconcentration Factor

The process by which chemical concentration within aquatic organisms exceeds that in the medium through dietary, respiratory, or other exposure routes in natural conditions. Bioconcentration is similar except here the intake of the chemical is entirely by respiration from the environment (Mackay et al. 2018). Bioaccumulation factor (BAF), on the other hand, is the ratio of the concentration of a chemical within an organism (C_O) to dissolved chemical concentration in water (C_W):

$BAF = C_O/C_W$.

Bioconcentration factor (BCF) can be defined as the ratio of the concentration of a chemical within an organism (C_O) accumulated solely by respiratory means to the concentration of that chemical dissolved in water (C_W):

$BCF = C_O/C_W$.

3.3.2 Bioamplification or Biomagnification and Biomagnification Factor

Bioamplification or biomagnification implies an increasing concentration of a chemical with increasing trophic levels in a food chain/web. Thus, this one is dependent on the diet of an organism so that the predators accumulate more of that compound. Biomagnification factor (BMF) is the ratio of the chemical concentration within an organism (C_O) to that present in its food (C_D) (Hoffman et al. 2002; Mackay and Fraser 2000; Mackay et al. 2018).

$BMF = C_O/C_D$.

Another important criterion for a chemical to be a potent bio-accumulator is its hydrophobicity and lipophilicity. The more hydrophobic a chemical is, the more is its bioaccumulation and biomagnification potential (Kelly et al. 2007; Walters et al. 2011). This property is determined using the calculation of the Octanol-Water Partition Constant/ coefficient or $K_{ow.}$ K_{ow} is the ratio of a chemical's molar concentrations between octanol and water phases (Hoffman et al. 2002).

$K_{ow} = [solute]_{octanol}/ [solute]_{water}$ (Square brackets indicate molar concentrations)

Chemicals that are poorly metabolized and highly hydrophobic with $K_{OW} \geq 10^5$ are particularly of much higher probability to biomagnify in an aquatic organism, and those with low K_{OW} are less potent (Kelly et al. 2007; Walters et al. 2011). However, further study has revealed K_{OW} is not the ultimate determinant in bioaccumulatory pathways. Especially in terrestrial food chains, less hydrophobic chemicals with $K_{OW} < 10^5$ and with low BCFs (regulatory criterion being 5000) were found to biomagnify (Kelly et al. 2007). Increasing evidence exists that emerging pollutants such as PhACs and EDCs are potent bioaccumulators in aquatic organisms (Ruhí et al. 2016). Pharmaceuticals are widely used, and hence there are several pathways including anthropogenic sources like hospital waste, human exertion, and wastewater treatment plant (WWTP) effluent, and so on that contribute to their build-up in the environment (Puckowski et al. 2016). Ruhí et al. (2016) studied bioaccumulation and bio-magnification effects of 25 PhACs and 12 EDCs in the aquatic ecosystem with three macroinvertebrates (*Ancylus fluviatilis*, *Hydropsyche sp*., and *Phagocata vitta*) up to secondary consumer level (Ruhí et al. 2016). Among all, the anti-inflammatory drug diclofenac, the lipid regulator gemfibrozil, and the flame retardant TBEP were detected in water, biofilm, and macroinvertebrate taxa. TBEP in particular showed distribution across all the taxa in the study with a considerable amount of magnification across trophic levels.

Perfluorochemicals (PFCs) used in heat, oil, stains, or grease-resistant products showed considerable accumulation and biomagnification across trophic levels starting from zoobenthos, phytoplankton to carnivorous fish, and egrets (top predators) in a food web in Taihu Lake in China. The magnification factors (TMFs) of perfluorooctane sulfonate (PFOS) was generally lower (2.9) than that of Perfluorinated carboxylates (PFCAs) (2.1 to 3.7) (Xu et al. 2014). The waste-water treatment plant effluents contain significant concentrations of antiepileptic drug carbamazapine (CBZ) into receiving waters. The bioaccumulation of CBZ was tested in an experimental trophic chain containing a green alga (*Pseudokirchneriella subcapitata*), a crustacean (*Thamnocephalus platyurus*), and a cnidarian (*Hydra attenuate*). It was found that CBZ did accumulate with BAF of 2.2 and 12.6, respectively, in *Pseudokirchneriella subcapitata* and *Thamnocephalus platyurus* but not in the cnidarian (Vernouillet et al. 2010).

Norsertraline, a metabolite of sertraline, an antidepressant drug, showed the highest BAF of up to about 3000 in the hepatic tissue of *Scardinius erythrophthalmus* in the North American Great lakes (Arnnok et al. 2017). In an experiment detecting bioaccumulation of pharmaceuticals in an aquatic food web consisting of damselfly and may fly larvae, water louse, ramshorn snail, and European perch hydroxyzine showed the highest BAF. In the food web, the ramshorn snail (a benthic species) and water louse demonstrated the highest concentrations, and oxazepam concentrations increased throughout the study in the perch (top-predator) (Lagesson et al. 2016). Fick et al. (2010) reported that among several other pharmaceuticals, the progestin pharmaceutical levonorgestrel was particularly of concern for its high bioaccumulation potential, as its plasma concentrations in fish blood even exceeded that of human plasma level (Fick et al. 2010). A higher concentration of the antidepressant sertraline and its metabolite desmethylsertraline was found in the Asian clam and two unionid mussel species. Although carbamazepine had a slightly positive TMF (1.17), the compound did not experience trophic magnification. Fifteen pharmaceuticals, including Sertraline, Desmethylsertraline, Orlistat, Tamoxifen, Ketoconazole, Promethazine, Diltiazem, Dextromethorphan, and so on, were found to be potent bio-accumulator for their high BCF with some of these even exceeded Human Therapeutic Dose (HTD). The concentration of antihypertensive diltiazem was higher in fish than in water and that in Osprey (a fish hunting raptor) was higher than the fishes, and this shows a magnification potential (Lazarus et al. 2015). Tamoxifen was found to be very bioaccumulative with BCF values as much as 12,000 and BAF up to 22,000 inside daphnids indicating significant impact of the dietary route (Orias et al. 2015). Diclofenac is an NSAID that is used extensively and so frequently present in aquatic environments. When tested for BCF of diclofenac in different fish tissues, it was found to have the highest range of BCF of 121 L kg^{-1} in the liver, 52.3 L kg^{-1} in the gills, and 46.8 L kg^{-1} in the muscle which allowed it to bioaccumulate in fish tissues (Lee et al. 2011). Kim et al. (2014) reported that the very commonly used antibiotic has bioaccumulation potential through aqueous or dietary uptake. Bioconcentration factors of 4.40 ± 0.91 L kg^{-1} were recorded for *D. magna* when it was fed upon contaminated algal food (Kim et al. 2014). Another commonly used antibiotic, roxithromycin, had a low BMF and still had BCF as high as 74.6 L kg^{-1} in the green algae *Scenedesmus obliquus.* Feeding on the contaminated algae resulted in accumulation in *D. magna* tissue. Similarly, the carp *Carassius auratus* when fed on contaminated *D. magna* tissue burden of roxithromycin was found to be in this order: the liver > muscle > gill and > bile (Ding et al. 2015).

Persistent organic pollutants (POPs) are chemically synthesized organic compounds that consist of organochlorine pesticides (OCPs) as well as polychlorinated biphenyls (PCBs) (Deribe et al. 2011). POPs, as their name suggests, persist in the environment for a very long duration and eventually gets deposited in the fatty tissue of animals as lipophilic compounds preferentially accumulate in stored lipids of animals (Mackay and Fraser 2000), and thus, fishes having high lipid content are prone to greater accumulation (Deribe et al. 2011). In a recent study by Deribe et al. (2011), DDT showed high bioaccumulation with great affinity in all the fish tissues from different trophic levels tested along with significant biomagnification across the food web, mainly in the fatty tissues of carnivores (Deribe et al. 2011). Another study involving heptachlor compounds, hexachlorocyclohexanes (HCHs) and DDTs and freshwater plants, and animals (shrimp and fish) showed that DDTs and heptachlor compounds were the strongest accumulators, followed by HCHs (Yu et al. 2012).

Flame retardants like Dechlorane Plus (DP) or polybrominated diphenyl ethers (PBDEs like BDE-28, −47, −66, −99, −100, −85, −153, −154, and − 183)) have TMF of much higher than 1 and thus highly bio-accumulative in nature in freshwater food webs that includes organisms of almost all trophic levels (Kannan et al. 2005; Pérez-Fuentetaja et al. 2015; Ruhí et al. 2016). POPs like DDEs, DDDs, and higher chlorinated PCBs (PCB 138, 153, and 180) tested TMFs between 1.5 and 4.2 indicating their high potential biomagnification along the given food chain. The extremely long half-life of such compounds (*p,p′*-DDE, and higher chlorinated-PCBs) within the organism is another noticeable fact (Kannan et al. 2005; Pan et al. 2017). Most of the PCBs found in freshwater systems have TMF higher than 1 as well as higher than that of PBDEs, proving their biomagnification potentials (Wu et al. 2008; Yu et al. 2012). It was also noticed that PCB bioaccumulation was positively related to higher trophic levels, proving its biomagnifying property (Kannan et al. 2001). Another emerging concern in the vast array of pollutants is microplastic pollution. Although having low bioaccumulatory potential, it is seen in many studies that trophic transfer of microplastics and nano-plastics is possible (Farrell and Nelson 2013; Ghosh et al. 2021).

The bioaccumulation and magnification potential of heavy metals is widely studied and very well known to scientific communities. Ample research works are available to support trophic transfer, bioaccumulation, and often biomagnification of metal and metallic products across different food webs in freshwater and marine water or terrestrial ecosystems. Among all the metals, although bioaccumulation was seen in several metals, Cd, Hg, and Zinc showed biomagnification potential, whereas for metals like As, Aluminium, Cr, Ni, Copper, or Lead it was relatively low and often declined with increased trophic level (Chen and Folt 2000; Jaishankar et al. 2014; Moloukhia and Sleem 2011).

Cd is one such toxic non-essential heavy metal that progressively accumulates in higher trophic levels in most of the food webs consisting of primary producers to predatory fishes in freshwater systems (Croteau et al. 2005; Cui et al. 2011). In a three trophic-level experimental food chain, it was seen that the first two trophic levels accumulated Cd with a similar higher magnitude than the later predatory trophic-level organisms. However, Croteau et al. (2005) showed Cd can increase or be biomagnified up to 15 times within two trophic levels. Another well-evidenced

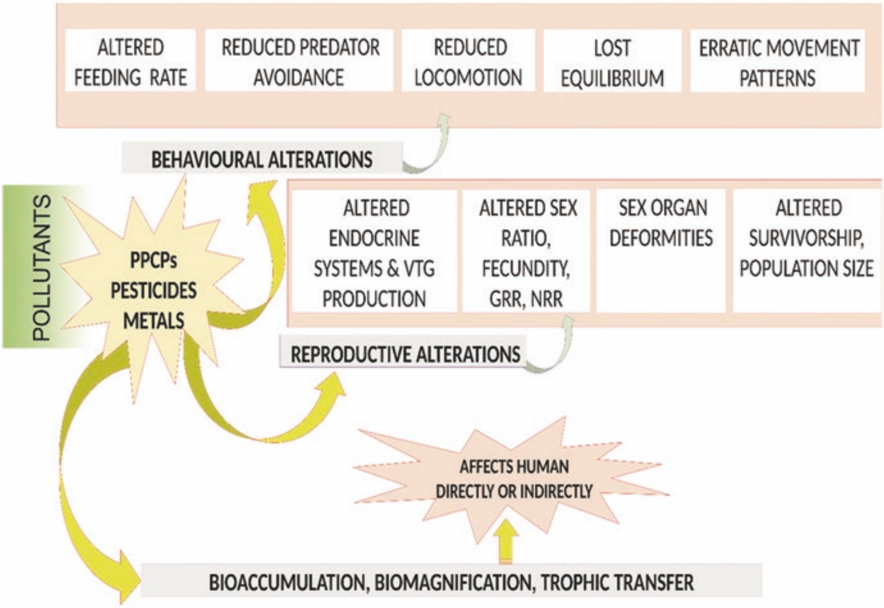

Fig. 10.6 Generalized model on the effects of different classes of riverine pollutants on the riverine and extra-riverine ecosystems

bio-accumulator and often biomagnification-capable metal is Mercury and its derivative MeHg (Croteau et al., 2005). In most cases, it was documented that MeHg was the more prevalent one in different trophic levels across several food webs than metallic Hg (Chumchal et al. 2011). In the biomagnification study of total and methyl mercury in Caddo Lake of the United States that included invertebrates, fish, as well as amphibians, reptiles, and mammals, it was demonstrated that MeHg was more accumulative with increasing trophic level, except for the lowest primary consumer level, where inorganic Hg predominated. This indicated that biomagnification did occur in the food web (Chumchal et al. 2011). Contrasting to many other studies, a study consisting of water, sediment, and zooplankton samples, it was seen that biomagnification of Pb through at least one trophic level was possible in freshwater systems as BCF of the predatory zooplankton was almost 4 times higher than the grazing ones (Rubio-Franchini et al. 2008). Accumulation of toxic components in aquatic organisms especially in the edible ones poses a great risk to humans, as these could be transferred directly via food (Jaishankar et al. 2014; Tchounwou et al. 2012). An overview of the noxious effects of various riverine pollutants on the riverine and extra-riverine ecosystem is depicted in Fig. 10.6.

4 Remediation Approaches: Advantages and Drawbacks

Considering the impact of riverine pollution on the riverine and extra-riverine ecosystems, several remediation approaches have been developed and attempted. This includes physical, chemical, and biological remediation strategies. Physical methods majorly involve filtration, adsorption, sedimentation, and so on. In this context, rapid adsorption using activated sludge, carbon blacks, powdered activated carbon, and pyrolyzed rice husk, following biodegradation are considered as cheap and environmentally benign approaches (Mukherjee et al. 2013). Application of electricity, that is, high-voltage pulsed discharge, has also been reported to be an effective strategy to remove organic contaminants.

Chemical remediation majorly involves the treatment of waste or polluted water using different chemicals that remove the pollutants through oxidation-reduction, hydroxylation adsorption, and so on. In general, the Photo-Fenton reaction is a widely used approach for removing various aromatic compounds (Mukherjee et al. 2013). The solvent-impregnated resin system is capable of forming a complex with various aqueous phenolic contaminants by hydrogen bonding and results in their removal (Mukherjee et al. 2013). Similarly, solid polymer electrolytes forming ion exchange membranes can efficiently detoxify wastewater through the electrochemical process (Mukherjee et al. 2013). In recent times, several types of nanomaterials including nanocomposites and nanoparticles have been exploited for remediation of inorganic and organic pollutants (Dey et al. 2018; Mukherjee et al. 2013).

Bioremediation is one of the most promising treatment strategies for removing pollutants from water by using microbes, plants, and their products. Phytoremediation by plants, microbiological remediation by bacterial and fungi, and enzymatic remediation majorly constitute the bioremediation approaches. Plants and microorganisms are used singularly or in synergism for the removal, degradation, or deactivation of pollutants (Bhat et al. 2020; Victor et al. 2016). Phytoremediation in itself is a combination of several other processes that together form the strategy: Phytosequestration or phytostabilization (a process in which pollutants are absorbed or adsorbed and immobilized in the vacuoles of roots); Phytoextraction (a process in which pollutants are taken up and stored in the tissues by the plants); Phytodegradation (the degradation of pollutants by the secreted enzymes by plants); Phytovolatilization (a process by which plants convert and discharge pollutants into a gaseous form into the atmosphere); and so on (Upadhyay et al. 2019; Victor et al. 2016). Several aquatic macrophytes like *Eichhornia crassipes* and *Pistia stratiotes, Glyceria maxima, Myriophyllum spicatum, Lemna gibba, Oenanthe javanica, Azolla caroliniana, Ludwigia peploides, Typha angustifolia, Typha latifolia, Vetiveria zizanioides, and so on* also showed great potential as phytoremediation agents for wastewater having high COD and BOD and contaminated with nitrogen congaing organic pollutants, ammonium, metals, PO_4^{3-}, PPCPs, pesticides, etc. (Mayo and Hanai 2017; Victor et al. 2016). On the other side, floating treatment wetlands (FTWs) provide good phytoremediation of mineral oil-contaminated wastewater. It was seen that *Brachiara mutica* and *Phragmites australis* when used together with bacterial strains *Bacillus cereus*, *B. amyloquefaciens*, *B. subtilis, B. licheniformis* (strains: LCRH93, BRRI53, LORI66, BRSI58, respectively) *Klebsiella sp. LCRI87* and Acinetobacter sp. (strains: TYRH47, BRSI56, CYRH21, LCRH81) showed better detoxification of the wastewater (Bhat et al. 2020; Rehman et al. 2018) Li et al. (2020) reported that constructed wetlands with plants such as *Typha angustifolia*, along with rhizosphere microbial communities efficiently, reduces different pollutants like organics, ammonia, phosphorus, and pharmaceutical like Ibuprofen in the wastewater (Li et al. 2020). The removal efficiency reaches a maximum at the bolting or mature stage of the plants. Constructed wetlands with plants like *Typha*, *Phragmites*, *Iris,* and *Juncus* remove PPCPs very efficiently, by themselves, or sometimes in synergism with rhizosphere and endophyte microbes. These microbes along with root exudates help in degrading the PPCPs in the medium (Nguyen et al. 2019; Zhang et al. 2016).

Several groups of microbes having various plasmids and catabolic genes and properties like hyper-mutation or mutational drift drive their extreme acclimatization power to adverse conditions like polluted medium, making them good bioremediation agents for many instances including POPs (Chakraborty and Das 2016). One of the best bioremediation techniques involves forming biofilms, an aggregation of microbes, algae, and fungi immobilized in a polymeric matrix. Biofilms containing *Bacillus* sp., *Escherichia* coli, *Pseudomonas* sp., and so on efficiently clear heavy metals from water by converting them to less toxic or unusable forms. Not only metals but biofilms are also exceptionally useful against PPCP, pesticides, hydrocarbons, and even highly resistant plastic pollution as well (Dar et al. 2020). Some commonly used microbes for the pesticide (bromophenol, chlorophenol, DDT, organophosphates, thiabendazole, Atrazine, etc.) remediation in different setups like biofilms include *Aspergillus* sp., *Ochrobactrum* sp., *Bacillus* sp., *Mycobacterium* sp., *Pseudomonas* sp., *Pandoraea* sp., *Rhodococcus* sp., *Klebsiella* sp., *Phragmites* sp., *Phanerochaete* sp., *Chrysosporium* sp., *Anthracophyllum* sp., and *Sphingomonas* sp. (Bhat et al. 2020; Dar et al. 2020; Girijan and Kumar 2020; Mondal et al. 2017; Mukherjee et al. 2013). *Cymbella* sp*., Navicula* sp*., Cocconis* sp*., Oocystis* sp*., Oedogonium* sp*., Aspergillus* sp., and others have shown potential degrading plastic wastes (Dar et al. 2020). Microbial remediation of metal pollution is also very effective as seen in many studies. Microbes like *Enterobacter* sp., *Pseudomonas* sp., *Achromobacter* sp., *Bacillus licheniformis*, *B. subtilis*, *Micrococcus* sp., *Escherichia coli*, *Acinetobacter calcoaceticus,* and so on have shown immense potential in the removal of metallic load (Dar et al. 2020; Mukherjee et al. 2013; Girijan and Kumar 2020).

Plant-microbe synergism acts great for the degradation of several pharmaceuticals like Carbamazepine (CBZ), Ibuprofen (IBU), Triclosan (TCS), and so on by plants, rhizosphere, and added endophyte microbes (Nguyen et al. 2019). Heterotrophic bacteria, symbiotic cultures of bacteria, and algae have also have proven their worth as remediation for PPCP pollution (Girijan and Kumar 2020). The use of different remediation strategies, their advantages, and drawbacks have been described concisely in Table 10.4.

Table 10.4 Different types of remediation approaches, their advantages, and disadvantages

Remediation type	Agents used	Advantages	Disadvantages	References
Physical remediation	Aeration	Enhances the growth and the vitality of micro-organisms	Not beneficial for toxic remediation	Wang et al. (2012)
	Sedimentation	Low cost and large pollutants are easily separated	Not applicable for dissolved pollutants and time-consuming	Yuan et al. (2018)
	Coagulation	Separates the suspended and dissolved compounds	A complex process, cost-effective, sludge is hazardous and requires costly disposal	Wang et al. (2012)
Chemical remediation	Peroxides	Low cost and availability, in-situ treatment, no by-products	Change in the pH of the water and habitat destruction on prolonged use	Romero et al. (2009)
	Sodium alginate (SA)-modified nanoscale zero-valentiron (NZVI)	Immobilizing, increase residual fraction, and decrease the bioavailability of cd	Long term effect is ineffective	Huang et al. (2016)
	Leaching	Removes heavy metals and results in shifting and cleaning the sediment	Time effective, cost-effective, and low chemical stability	Cui et al. (2011)

(continued)

Table 10.4 (continued)

Remediation type	Agents used	Advantages	Disadvantages	References
Enzymatic remediation	Hydrolytic enzyme	Easy availability, lack of cofactor stereoselectivity, and tolerance to water-miscible solvents	Economically high procedure	Mukherjee et al. (2013)
	Polyphenol oxidase	It does not require H_2O_2 and low-cost treatment	A bulk amount is required for complete remediation. Enzyme inactivation and purification cost which limits its industrial applicability	Mukherjee et al. (2013)
	Oxidoreductases	Effective for heavy metal, phenol, and PCB transformation	Lose their activity upon pollutant transformation	Gianfreda and Rao (2004)

Remediation type	Agents used	Advantages	Disadvantages	References
Phytoremediation	Phytoextraction- *Typha latifolia, Sida cardifolia, Chenopodium album*	Cost-effective, recovery, and re-usable. Easily monitored "green" remediation technique	Slow hyperaccumulation and improper disposal of the biomass	Gupta and Sinha (2008)
	Phytodegradation/phytotransformation by *Daucus carota, Festuca arundacea, Lolium multiflorum, Amaranthus sp. Raphanus sativus, Ipomea sp., Sorghum drummondii, Chenopodium rubrum*	Plant enzymes mediated the degradation of organic compounds within roots or leaves to simple molecules	Large-scale use cause-effect on fauna diversity	Gajić et al. (2018)
	Eichornia crassipes and *Lolium perenne,* aquatic macrophyte *Eleocharis acicularis, Phragmites, Lemna, Eichchornia, Azolla, and Typha*	Hyperaccumulator of heavy metal; cost-efficient, lower technology requirement, and the cleaned product can be used as fertilizer, forage, mulch, or for the production of biogas	Inhibit the growth of algae and bacteria, block sunlight to reach the bottom. Time taking and large-scale infrastructure is needed	Bhat et al. (2020), Dar et al. (2020), Kumar and Bharadvaja (2020)
Microbial remediation	Microbial fuel cell- *Geobacter sp.*	Bioelectricity generation and wastewater treatment	Slow rates of substrate degradation	Sevda et al. (2013)
	Bacillus subtilis, Alcaligenes sp., Ralstonia eutropha, Sphingomonas sp., and Nocardioides sp.	Degrade chlorophenol, trichlorophenol, and tetrachlorophenol	Not effective for large-scale use	Mukherjee et al. (2013), Rehman et al. (2018)
	Fungi-based remediation- *Trametes versicolor, Pleurotus ostreatus, Aspergillus wentii,* and *Aspergillus Niger.*	Mycological bioremediation of phenols	Sensitive to the level of toxicity and time ineffective	Mukherjee et al. (2013)
	Bioaugmentation technology using *bacillus* and *Rhodobacter,* biofilm technology	Reduce significantly COD abundance and cost-efficient	Cannot help in denitrification	Yuan et al. (2018)

5 Conclusion and Future Directions

Considering the alarming stipulation of riverine pollution and its effect on the biosphere, the present chapter illustrates a generalized overview of the different chemical classes of pollutants, their effect on the ecosystems as well as a molecular mode of action and possible route of remediation. Presently, the remediation strategies have been focused on two critical aspects, namely, biomonitoring/biosensing and biodegradation. Oxido-reductive enzymes from plants and microorganisms are being used for both the detection and the treatment of organic and inorganic contaminants from water. Polyphenol oxidase (PPO) from bacteria and plants is considered a cheap enzyme with excellent detection and biodegradation capacity against aquatic phenolic contaminants (Mukherjee et al. 2012, 2013). Bacteria belonging to the genus *Acinetobacter* have been reported to be involved in removing surface As contaminants (Goswami et al. 2015). Recently, in situ sensing and degradation of phenolic compounds have been reported for PPO immobilized on gold-doped graphene oxide nano scaffold (Mondal et al. 2017) and luminescent hydrogel (Dey et al. 2018). However, more research studies are particularly needed in this field especially in detecting and removing recalcitrant toxicants. The Discovery of new bioindicators that will discriminate between healthy and polluted riverine ecosystems is of major importance. Finally, global awareness through mass campaigning is also necessary to fight this battle not only for the riverine organisms but also for all other ecosystems that are connected directly or indirectly with the riverine ecosystem.

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Chapter 11
Assessment of Habitat Quality in Quarried Reach of Alluvial River

Raj Kumar Bhattacharya, Nilanjana Das Chatterjee, and Kousik Das

Abstract Fluvial ecosystem function and biological communities entirely depended on habitat quality; however, increasing anthropogenic intervention damages the habitat quality day by day. Hence, this study has attempted to assess the impact of anthropogenic intervention mainly sand mining on three-tier habitats, that is, river bed, riparian and bank in Kangsabati River, West Bengal, India. Channel geometric, geomorphic responses, sediment facies and their deposition, riverine land use, microhabitat zone, water quality, species diversity were considered as components for habitat quality assessment (HQA). Quality score is estimated from the relative weightage of HQA indicators in selected habitat sites. Result demonstrated that maximum mining and pit sites in three-tier habitat systems fall under marginal (2–4) and poor category (<2), while sandbar sites in habitat system reach optimal (>4) and suboptimal (3–4) category, respectively. Mining affected channel width drastically reduces but channel depth sharply increases through the generation of turbulence flow, pool-riffle alteration, thalweg shifting, which are causes of loose coarser deposition (LCD) in bed, sand clay deposition (SCD) in riparian, and medium finer deposition (MFD) in a bank. Moreover, the mixing of pebbles with MFD disrupted the herbs and rubiaceas grass colony in the bank due to pit generation, while interruptions of SCD damage the grasses colony in riparian sites to enhance the harmful land use practice. Contrastingly, physicochemical properties continuously deteriorated in mining and pit sites; thus, species diversity and richness relatively declined but species evenness abruptly increased. Therefore, habitat quality has gradually degraded from bed to bank sites of mined rivers.

Keywords Habitat quality assessment (HQA) · Turbulence flow · Loose coarser deposition (LCD) · Sand and clay deposition (SCD) · Medium finer deposition (MFD)

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1 Introduction

Physical habitat in riverine ecosystem denotes the living space of instream, riparian, and bank biota, which has greatly temporal and spatial dynamic entities that are determined by the several interactions of structural features between channel geometry and hydroecological regime (Barquin Ortiz and Martinez-Capel 2011; Yu et al. 2019; Yang et al. 2018). Habitat realms are considered as wide ranges from small microhabitat levels to large catchment realms, where species can grow indigenously in response to hydraulic variables like substrate composition, water depth, flow velocity, and ecological variables like water quality, nutrient density in the entire channel (Dutta et al. 2017; Yang et al. 2018). Moreover, the topographical features of the instream and floodplain site not only controlled the habitat configuration but also directly determined the aquatic community (Adaland 1993). Generally, habitat ecosystems progressively developed with the surrounding biotic, abiotic components and adjacent environment. As a result, habitat ecosystems can successfully evaluate the river health to develop throughout the river, which provided the natural link between all inhabitants and its adjacent environment (Galbraith et al. 2016; Maddock 1999). Contrastingly, overgrowing anthropogenic activities, that is, navigation, agricultural irrigation, water pollution, dam construction, sand mining, and channelization have drastically deteriorated the global riverine system including water quantity and water quality. In this context, hydroecological and physical attributes of the channel inversely altered the entire composition of the physical habitat; thus, the biodiversity of river ecosystems highly degraded day by day (Galbraith et al. 2016; Yang et al. 2018). This degradation is widespread in the aquatic ecosystems; subsequently, species population in the freshwater aquatic communities reduced by up to 81% since 1970 as estimated by Freshwater Living Planet Index (WWF 2016). Therefore, freshwater habitat ecosystem faced more threats and also hosted disproportionally a huge number of dominant species communities to consider their worldwide rarity (Gothe et al. 2015; Strayer and Dudgeon 2010). On the other hand, freshwater biodiversity is facing underappreciated threats caused by overgrowing and pervasive fragmentation of river systems. This fragmentation is initiated by two ways, that is, natural and anthropogenic, where natural ways are generated by different agents like a waterfall, cascade, but anthropogenic ways are developed by human actions like damming, road building, water lifting, and water quality barriers (Fuller et al. 2015).

Assessment of physical habitat is one of the most significant approaches in fluvial geomorphic studies for determining the key roles of channel geometry and hydroecological attributes of physical habitat realms on one hand, and to accurately analyze the various processes engaged for generating heterogeneity spatial entities as well as specific physical features on the other hand (Barquin Ortiz and Martinez-Capel 2011; Butler 1995). Recently numbers of effective indicators and specific methods have been adopted to measure the physical and hydroecological attributes of riverine habitat ecosystems in the entire world (X. Yu et al. 2019). In particular, the index of Habitat Integrity (IHI) is one kind of effective habitat assessment method, which measures the integrity between instream and riparian zone with the following of hydrological, physicochemical, river bed, and all their connective modifications (Kleynhans and Louw 2008; Yu et al. 2019). River Health Programme (RHP) was initiated by the Forestry of South Africa and the Department of Water Affairs for assessing the adopted responsibility of instream and riparian biological communities to various disturbances in an aquatic environment (RHP 1994). Based on the spatial scale of habitat realms, three types of methodologies are applied, that is, broad-scale assessment method, microhabitat assessment method, and empirical habitat assessment method (Yang et al. 2018). Broad-scale assessment method is used for classifying the entire river systems into small scale sub-segments including their types and spatial situation, ranges from large drainage basin scale to sub-segment reach scale. This classification is based on various physical attributes like channel pattern, stratigraphic situation, channel bed slope, hydraulic regime, surrounding land use patterns, as well as long-term historic and field investigation data. While microhabitat assessment method is mainly involved to determine the various small scale hydro-morphological attributes, that is, water depth, flow velocity, substrate composition, riparian land use pattern along with biotic composition and its structure, ranges from sub-segment reach scale to patch scale (Sarremejane et al. 2017). Empirical habitat assessment method is applied with the development of regression models to predict the bio-physiological status ranges from small reach scale to patch scale using field investigation database and empirical determination techniques such as HABSCORE (Milner et al. 1985) and habitat quality index (HQI) (Binns and Eiserman 1979). Moreover, empirical habitat model is actively used for river restoration and management at the basin (Ding et al. 2015; Yang et al. 2017) and urban river scale (Mosner et al. 2015; Choi and Choi 2015) due to insufficient historical database regarding the river habitat.

In India, many scholars are attempted to assess the physical habitat using various methods such as the overlay method applying on Denwa river (Vyas et al. 2013), Ganjal and Morand River (Sharma et al. 2018). Present study is selected on Kangsabati River in south Bengal, India, caused by various anthropogenic activities like sand mining, dam construction, and other natural river network fragmentation creates more hindrances on river habitat over the years (Bhattacharya et al. 2020). Therefore, habitat loss or degradation is a challenging issue in this alluvial river. Based on these entire literature, two major gaps are identified on habitat assessment at sub-segment to patch level, that is, not signify the separate role of individual variables in a channel and ecological attributes on three-tier habitat ecosystem as channel bed, riparian, and bank sites, and not mention the correlation between hydroecological succession with habitat parameters. In respect to the above research gap, the assessment rank score scale aims to (1) assess the status of channel and ecological attributes in channel bed, riparian, and bank sites, (2) identify the correlation between channel attributes and ecological attributes on growing hydroecological succession in three-tier habitat system and (3) find out the threshold value of a more effective variable for further reaching the resilience state to manage the river restoration at patch scale.

2 Methodology

2.1 Study Area

The Kangsabati is a tropical alluvial river flowing along the south-easterly direction in accordance with water feeding; this river flow is regarded as the last contributing river to the Ganges River in India. Kangsabati River originated in the up elevated lands of Chhota Nagpur plateau under Jharkhand, then passing through Purulia, Bankura, Paschim Midnapore, and Purba Midnapore districts of West Bengal. Kangsabati River is generated from Kansai and Kumari River near Mukutmonipur dam, then flows toward the downstream with the joining of such tributaries like Taraphini and Bhairabbanki at Sijua near Lalgarh. During downstream flow, Kangsabati River is divided into two separate branches of new Kansai and old Kansai, where new Kansai join the Rupnarayan River at Palaspai while old Kansai joins the Kheliaghai River near Panskura. This non-perennial river extended by latitudinally from 21°45′N to 23°30′N, and longitudinally from 85°45′E to 88°15′E with covering an area of 9658 km^2 (Fig. 11.1). Most of the eroded and gullied landscapes are observed in upper catchment sites where dominant first and second stream orders make draindatic and sub-draindatic drainage patterns. The average rainfall in the upper catchment ranges from 1080 mm to 1200 mm while rainfall in

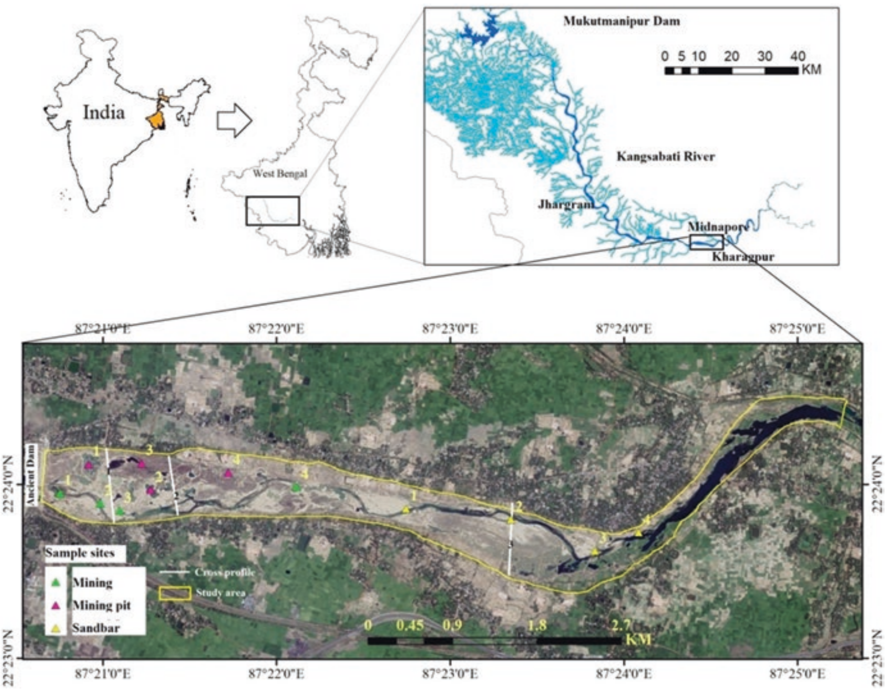

Fig. 11.1 Study area

the middle and lower catchment ranges from 1500 mm to 1600 mm. Physical habitat systems of floristic realms have been identified in three different sites, that is, riparian, bank and shaded aquatic (Bhattacharya et al. 2020). Floating and submerged micro plant species name *Vallisneriaspiralis, Potamogetonnodosus*, *Marsileaminuta*, *Egeria spp*. (water weeds), *Mosquito fern* are colonized under aquatic ecosystem on channel bed whereas *Cynodondactylon, Mikania micrantha, Ludwigia perennis* live in riparian sites where moist sand and clay layer are more dominated. Contrary, *Koeleriamacrantha, Vetiveriazizanioides, Solanumnigrum* grow up healthy across the left and right bank sites with the presence of dry sandbar and weathered rocky outcrops. In this study, the study area is selected from Anicut dam to Paikpara along the Kansai River where latitudinal extension started from 22°23′30″ N to 22°24′00″ N and longitudinal extension as 87°21′ E to 87°22′30″ E. Three-tier habitats are gradually extended but in the recent past, many anthropogenic activities such as bricks production, sand mining has hampered this habitat ecosystem (Bhattacharya et al. 2019a).

2.2 Methods and Database

Current study was conducted and consulted with many research works on the assessment of physical habitat in the riverine ecosystem in published and unpublished literature and reports. Thus, secondary databases such as high-resolution Google Earth during 2002 and 2016, and Survey of India Toposheet maps as 73 N/7, 73 N/8 of 1:50,000 were taken to measure the land use land cover patterns (LULC), microhabitat zoning, riparian vegetation patch in the entire study area. Moreover extensive fieldwork was carried out to find out the registrar sand mining sites, mining pit sites, sandbar, different aquatic patches, fragmented riparian vegetation during post-monsoon season of 2016. Furthermore, Germin GPS (78CSx) and Total station were used to measure the three cross profiles, and several geomorphic responses, that is, pool-riffle sequence, thalweg line, sandbar deposition, sediment movement in nine different sites, that is, sandbar dominated bed, mining affected bed, pit form bed, sandbar dominated riparian, mining affected riparian, pit dominated riparian, sandbar dominated bank, mining affected bed, pit affected bank during 2016.

2.3 Assessment of Geomorphic Indicators

2.3.1 Estimation of Planform Change

Geomorphic attributes of river bed habitat are analyzed considering channel planform change across the different cross profiles during a long-term span between 2002 and 2016. Three indexes of sinuosity index (P), braiding index (Bi) and braiding channel ratio (B) were used to determine the channel planform change and its associated biophysical habitat system following Friend and Sinha's method (Friend and Sinha 1993).

Sinuosity Index (P)
This index is measured the meandering distance along the channel in the two years 2002 and 2016 as follow (Ghosh et al. 2016):

$$p = \frac{L_{\max}}{L_R} \quad (11.1)$$

where L_R indicates straight distance in accordance to thalweg line, $L_{c\max}$ denotes curvature distance following the thalweg line along with considered as mid-channel length from a single or multi-channel or widest channel.

Braiding Index (Bi)
Channel braiding index estimated the channel braiding through the measuring of braiding conversion into single-channel flow as well as channel widening initiating by anthropogenic and natural fragmentation (Bhattacharya et al. 2019a).

$$B_i = \frac{L_{\text{ctot}}}{L_{c\max}} \quad (11.2)$$

$L_{c\max}$ indicates curvature length along the mid-channel segment of the widest channel in this study reach; L_{ctot} denotes the measurement of thalweg line length along the main channel.

Braid-Channel Ratio (B)
Bank erosion and channel incision-induced riparian habitat change are detected using of braid-channel ratio as computed by coupling Eqs. (11.1) and (11.2) as follows (Bhattacharya et al. 2019b):

$$B = \frac{L_{ctot}}{P \times L_R} = \frac{P_{\text{cot}}}{P} \quad (11.3)$$

2.4 Assessment of Hydroecological Indicators

2.4.1 Water Sampling Process and Adopted Methods

Nine water samples were collected from three sites of sandbar-dominated riffle sites, pit-dominated pool sites, and sand mining sites in this selected study reach during 2015–2016. Based on ingredient composition, physicochemical properties in each water sample were variables from one season to another season. Every three samples were repeatedly collected in clean 100 g plastic bottles from the sandbar, mining, and pit sites during pre-monsoon, monsoon, and post-monsoon season throughout the year (2015–2016). After the collection of water samples, all bottles were washing with 15% HNO_3 (v/v) repetitively two or three times, and consequently, double distilled water was used to fill the bottles. Then all samples were filtered through the using of Whatman-542 filter paper that is prepared by G.E. Healthcare, U.K. LMT., and stored in a cold place at 4°C within a day (24 h) (Bhattacharya et al. 2019c). Several physicochemical parameters (PP) such as $_P{}^H$, TDS, turbidity, DO, BOD, salinity, Mg^{2+} were determined using water quality measuring tools, that is, Aqua Read Ltd., U.K of GPS Aqua-AP-1000. Furthermore, alkalinity in the water sample was measured using of auto-titration method (Microohm). Lastly, every PP of water samples was calibrated with the taken of rapid calibration solution (RCS) to get the accurate measurement either different sites or during different seasons. Based on the measurement database of PP, the descriptive statistical technique was applied to compute the range, mean, maximum, minimum, standard deviation, standard error, and variance for assessing the integration of water quality with instream biota under the SPSS platform (version-22).

2.4.2 Estimation of Water Quality Index (WQI)

Water quality index (WQI) is estimated from PP of water samples through the assigning of weightage values based on their relative importance under the weight-assigning technique. Highest weight value was assigned as 5 for more significant or maximum influence PP whereas the lowest weight value was assigned as 1 for less significant or minimum influence PP of water quality (Sharma et al. 2014; Bhattacharya et al. 2019c). The first step is computed for the relative weight (R_w) as follow:

$$R_w = A_w \div \sum_{k=0}^{n} A_w \tag{11.4}$$

A_w denotes weight assessment of every PP, n means number of considering PP for assessment of water quality. In this study, R_w value of nine samples was assigned from eight PP following Alobaidy's relative scale, that is, conductivity (5), DO (5), TDS (4), $_P{}^H$ (4), salinity (4), BOD (2) and Mg^{2+} (2) (Alobaidy et al. 2010).

Next step is involved mainly to compute the quality of rating scale (Q_{rs}) in each PP through the dividing of their standard values given by WHO and BIS scale follows:

$$Q_{rs} = \left[C_i \div S_i\right] \times 100 \tag{11.5}$$

On the contrary, various methods are applied to estimate the Q_{rs} value of DO and $_{P}{}^{H}$, although the measuring value of DO and $_{P}{}^{H}$ are deducted from their ideal value as 7 ($_{P}{}^{H}$) and 14.6 (DO) (Sharma et al. 2014).

$$Q_{rs}, P^{H}, DO = \left\{\left(\left(C_i - V_i\right) \mathrm{l} \left(S_i - V_i\right) / 1\right) \times 100\right\} \tag{11.6}$$

Q_{rs} means quality of rating scale, C_i presents concentration measurements, S_i denotes BIS scale base standard value of drinking water.

Finally, sub-indices (SI) of nine samples were computed to obtain the WQI based on Eqs. (11.7) and (11.8) (Sharma et al. 2014)

$$\mathrm{SI} = R_w \times Q_{rs} \tag{11.7}$$

$$WQI = \left[\int SI\right] \tag{11.8}$$

2.5 *Measurement of Microhabitat Patch*

Micro habitat patch in three-tier riverine habitat systems has been accounted using several biodiversity indexes such as Simpson index of dominance (1949), Shannon-Wiener index of diversity (1963), Marglef index of species richness (1958), Pielou index of species evenness (1966) (Bhattacharya et al. 2019c). In this study reach, 45 species were collected from the river bed, riparian and bank sites during pre-monsoon, monsoon, and post-monsoon season, respectively.

Simpson index of dominance: Species dominance and their relative abundance across the river bank to bed have been computed by Simpson index (D) (Shannon 1949) as follow:

$$D = \left\{\frac{N\left(n-1\right)}{N\left(N-1\right)}\right\} \tag{11.9}$$

where N denotes the total number of all living species in the entire study reach, n indicates the number of organisms in each species.

Shannon-Wiener index of diversity: Species diversity (α) in suitable three-tier habitat community ecosystem is determined by Shannon-Wiener index (H) using Shannon and Weaver (1963):

$$H = -\sum p_i \log p_i \tag{11.10}$$

where $p_i = S/N$; S indicates the individual number of specific species, N means the entire number of all individual species in every sample site, and IN demonstrated logarithm value to base e This index ranges from 0 to 1, where 0 reveals the absence of no diversity that means only one habitat patch lives there (Shannon and Weaver 1963).

Marglef index of species richness: Species richness means number of species that appear in each sample site. Species richness is measured through the considering of selected species number, which is divided by square root value in every individual of a sample (Bera et al. 2020). Marglef index (SR) is applied to determine the species richness in this alluvial stream reach as follows (Margalef 1958).

$$\mathrm{SR} = \frac{S-1}{in\,N} \tag{11.11}$$

Pielou index of species evenness: Species evenness is an important biodiversity index to effectively measure the relative abundance in different species across the river bed to riparian and bank using the Pielou index (Pielou 1966)

$$e = \frac{H}{InS} \tag{11.12}$$

2.6 Final Rank Score of Assessment Indicators

Habitat assessment indicators, threshold range, and resilience capacity are assessed with the applying of rank score techniques. Threshold range of geomorphic assessment indicators is generally based on effective indexes, that is, P, Bi, B; sediment textural attributes; geomorphic responses; and anthropogenic activities induced riverine land cover change. While threshold range value of hydroecological assessment indicators is determined by biodiversity indexes and water quality index as prescribed by WHO and BIS. Habitat quality score is determined with the following priority rank using the compound factor (CF). Thus, CF is calculated from Eq. (11.13) as follow (Altaf et al. 2014):

$$\mathrm{CF} = \frac{1}{n}\sum_{i=1}^{pn} R \tag{11.13}$$

where R means parameter rank value, and pn indicates parameter rank from each variable.

Therefore, rank 5 is giving from the highest value of all positive relationship variables, and the next rank is rated by the second highest value of this same relationship. Based on CF values, condition category is classified into four classes, that is, optimal (10–7), suboptimal (7–5), marginal (3–5), and poor (<3), respectively. Optimal and suboptimal categories are demonstrated that habitat assessment indicators are able to reach resilience state, while marginal category indicates the effective indicators are meet to threshold state. Moreover, poor category denotes that habitat indicators are crossed the threshold state, simultaneously several consequences would be arise throughout the channel.

3 Results and Discussion

3.1 Analysis of Geomorphic Indicators

Several indicators like the cross and long profile for channel geometry, sedimentary facies and textural composition for sediment grain size distribution, pool-riffle sequence and thalweg shifting for geomorphic responses, sinuosity index, braiding index and braid channel ratio for planform change, identification of geomorphic landform for measuring of natural and anthropogenic induced interruption are considered to analysis the threshold range and generating consequences of geomorphic attributes on three-tier habitat system throughout the study area. Based on the rank score of these geomorphic indicators, four habitat conditional categories are measured from river bed to bank margin.

3.1.1 Channel Geometry

Channel morphological attributes such as channel width, wetted width, bank height, channel depth, thalweg location, and its shifting trends are measured with the taken of three cross section (Yang et al. 2018) at AB (1), CD (2), EF (3) from left to the right bank, and one long profile (1) from Anicut dam to Debangai. Maximum channel bank width is observed at CD profile (830 m) than AB and EF profile, but average width as 650 m. While average wetted width ranges from 320 m to 180 m during 2016 (Fig. 11.2a, b, c), but highest bank height is observed at CD (8 m) and AB (7 m) profile where left bank integrated with extensive riparian zone; however, channel flow concentrated along the deepest thalweg line near the right bank. While riparian zone gradually expanded at the right bank, and channel flow with the deepest thalweg line concentrated near the left bank at the EF profile.

Consequently, bank erosion and collapsing occurred at the deepest channel flow bank side than the extensive riparian bank side. On the other hand, long profile demonstrated that both the thalweg line and channel flow frequently shifted due to the deepest instream mining pit sites along the bed. It is to be seen that habitat suitability has always grown up across the extensive riparian widen bank than the narrow bank. All geometric parameters are assigned as quality scores in different conditional categories with the following of Yang et al. 2017, 2018; Bhattacharya et al. 2019b (Table 11.1).

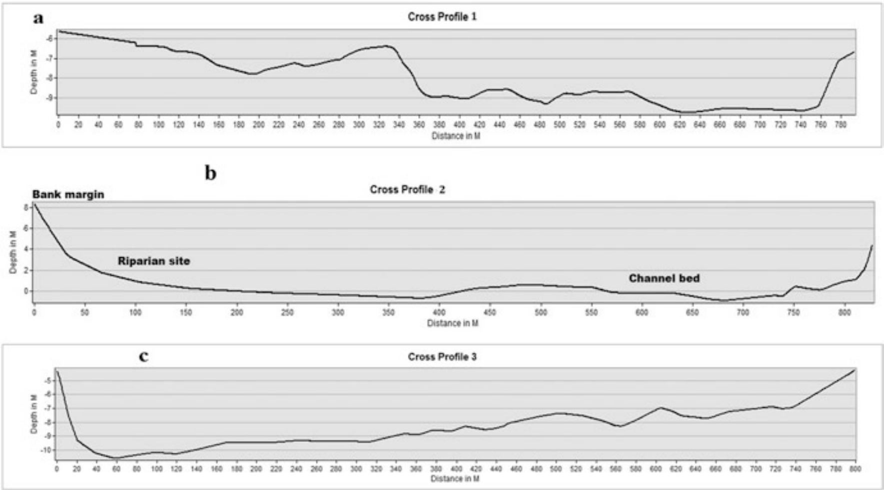

Fig. 11.2 Cross profile: (**a**) AB (1), (**b**) CD (2), (**c**) EF (3)

3.1.2 Sedimentary Facies and Textural Attributes

Sedimentary facies depict the substratum textural composition in different habitat (Yang et al. 2018). Three different sedimentary facies, that is, river bank, riparian, bed in Kangsabati River is presented in Fig. 11.3a, b, c.

Sediment Facies at Bank Margin Site

Based on depth variation and textural composition, six sedimentary layers are observed in the bank facies named as finer curve (0.1 m), clay finer (0.3 m), medium sand (0.1 m), coarse sand (0.25 m), medium silt (0.2 m), and clay finer texture (>0.3 m), respectively (Fig. 11.3a). Climb, hubs and grass are dominantly observed in finer curve layer while healthy grass root colony has been well established in clay finer layer due to supply of moisture content. Moreover, *Koeleriamacrantha, Vetiveriazizanioides, Solanumnigrum* including grassroots are extended up to medium and coarse sand layers, but the clay finer layer has only allowed the aquatic species to dominate along both bank margins.

Sediment Facies at Riparian Site

Six sedimentary layers have been identified in the riparian facies, that is, yellowish brown–based compact finer texture (0.1 m), white blackish sandy clay-based compact finer texture (0.15 m), brown–yellow based compact finer texture (0.2 m), brown white-based loose finer texture (0.12 m), brown blackish based compact finer

Table 11.1 Channel geometric impact on riverine habitat system and score assignment

Condition category	Channel geometry impact on hydroecological status	Score	Observation site
Channel width (Yang et al. 2017, 2018; Bhattacharya et al. 2019b)			
Optimal	Wide channel width; braided pattern; high width depth ratio; low mining and channelization	4	Sandbar sites
Suboptimal	Moderate channel width; meandering bend; medium width depth ratio; the presence of mining and channelization	3	Pit sites
Marginal	Narrow channel width; sinuous pattern; low width depth ratio; intensive instream mining	2	Mining sites
Poor	Steeply and narrow width; straighter pattern; low width depth ratio; intensive mining	1	Mining/pit sites
Channel depth (Yang et al. 2017, 2018; Bhattacharya et al. 2019b)			
Optimal	Low depth with slow-shallow; fast shallow; small depth pool	3	Sandbar sites
Suboptimal	Moderate shallow but absent of fast shallow; shallow pool	2	Sandbar sites
Marginal	Depth increases and completely missing of shallow depth; mining-induced turbulence flow creates pit pools	1	Mining sites
Poor	More depth and majority of pools with large depth	1	Pit sites
Riparian width (Bhattacharya et al. 2019b)			
Optimal	Width > 18 m; no instream and floodplain mining	4	Sandbar sites
Suboptimal	Width 12–18 m; moderate floodplain mining	3	Mining sites
Marginal	Width 6–12 m; intensive floodplain	2	Mining/pit sites
Poor	Width < 6 m; narrow riparian caused by massive floodplain sand mining	1	Pit sites
Bank stability (Bhattacharya et al. 2019b)			
Optimal	Stable bank; absence of bank erosion and shifting; potentiality of a future problem as <5%	4	Sandbar sites
Suboptimal	Moderate stability including least shifting; rate of bank erosion as 5–30%	3	Pit sites
Marginal	Moderate unstable and bank shifting; rate of bank erosion as 30–60%; mining-induced bank erosion has potentiality	2	Mining/pit sites
Poor	Unstable bank erosion and shifting; erosion scars are covering as 60–100%; massive bank sloughing near mining sites	1	Mining sites

texture (0.08 m), and brownish loose finer texture (>0.25 m), respectively (Fig. 11.3b). In terms of microhabitat patch, *Cynodondactylon, Mikania micrantha, Ludwigia perennis* are grown on yellowish-brown compact finer texture while grass rock colony including dry and dead rock colony are concentrated in brown white-based loose finer texture layer. It is to be noted that most of the lower layers in riparian sites are fully free from vegetation cover due to the presence of huge loose finer sand.

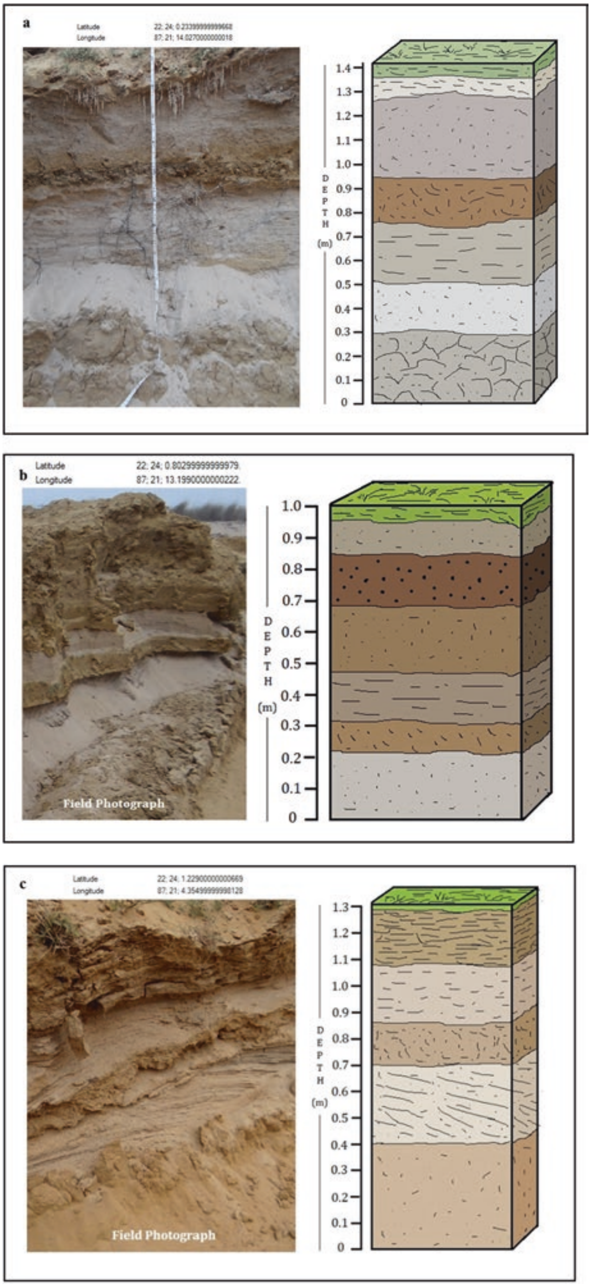

Fig. 11.3 Sedimentary facies: (**a**) river bank, (**b**) riparian, (**c**) bed

Sediment Facies on a River Bed

Five sedimentary layers have been identified on the river bed facies, that is, medium coarser sand (0.2 m), medium sand (0.175 m), finer sand (0.12 m), finer silt (0.25 m), and finer clay (>0.4 m), respectively (Fig. 11.3c). Maximum aquatic species like *Potamogetonnodosus*, *Marsileaminuta*, *Egeria spp*. live on the medium coarser and medium sand layer and some zooplanktons breathe finer silt and clay layer throughout the channel. In addition, aquatic species such as water weeds, *Mosquito fern* grow in mining pits due to the supply of healthy nutrients. It is true that aquatic organisms are fragmented and fragile in mined channels caused by excessive sand mining. Based on hydroecological status, sedimentary facies and sediment deposition are assigned the quality score in Table 11.2 following Bhattacharya et al. 2020.

3.1.3 Pool-Riffle Sequence

Pool-riffle sequence means undulating river bed surface involving depression and mound along the longitudinal stream caused by over extraction of sand from instream and floodplain and irregular sedimentation over the channel (Ghosh et al.

Table 11.2 Sedimentary facies and textural attributes impact on riverine habitat system and score assignment

Condition category	Sedimentary facies and textural attributes impact on hydroecological status	Score	Observation site
Sedimentary facies (Bhattacharya et al. 2020)			
Optimal	Clay finer and medium silt composed layer supplied moisture content; healthy vegetation patch; no prospect of floodplain sand mining	4	Sandbar sites
Suboptimal	Medium to finer sand layer supplied moderate moisture content along with little floodplain mining; sparse vegetation cover	3	Mining sites
Marginal	Coarse to medium sand layer reduces the moisture supply; mining hampered the gradation of sediment facies; degraded grass colony	2	Mining/pit sites
Poor	Mining affected sediment facies; mixing of sand, silt, and clay; absence of vegetation patch	1	Pit sites
Sediment deposition (Bhattacharya et al. 2019b, 2020)			
Optimal	Low gradient stream (<20%) and little expanded of point bars; small deposition affected on channel bed (<5%)	4	Sandbar sites
Suboptimal	Newly generation of sand bar composed with medium and finer sand due to low gradient (20–50%)	3	Mining sites
Marginal	Moderate deposition of sand for low gradient (50–80%) but pools are increasingly caused by mining	2	Pit sites
Poor	Massive deposition of finer and loose sediments; however, mining-induced pools are shifted in riffle; channel bed gradient interrupted	1	Mining/pit sites

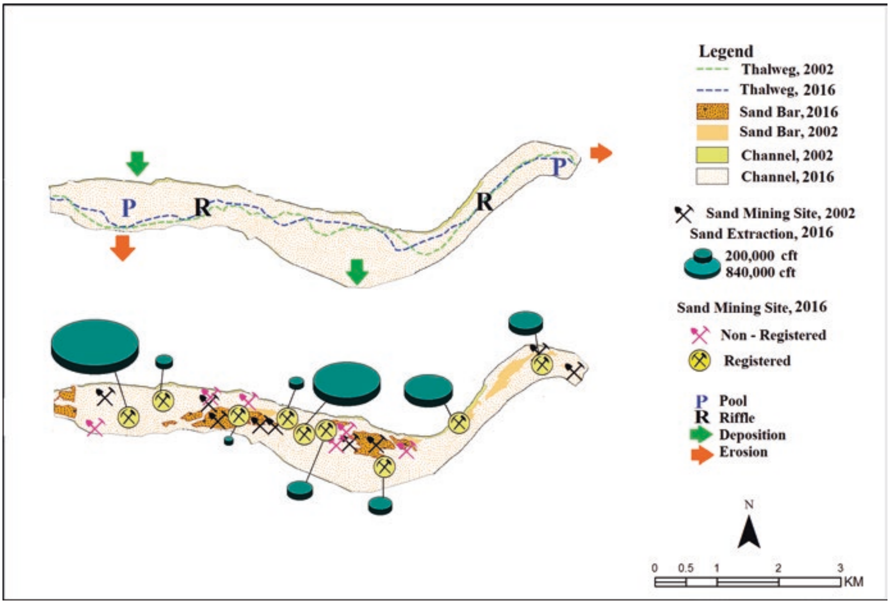

Fig. 11.4 Channel planform and sand mining intensity during 2002–2016

2016; Bhattacharya et al. 2019b). Contrastingly, pool-riffle sequences have greatly influenced in creating mesohabitat structures and aquatic ecosystems supplying healthy nutrients (Mg^{2+}) and PP (BOD, DO, TDS, etc.) (Calderon and An 2016). In this study, pool-riffle alteration is a very common hydraulic act to lead to the interruption of erosion and deposition process in the entire channel. Maximum pools are generated in sand mining sites (pit) after that channel joins in this depression, then in the channel, flow is propagated toward the downstream (Fig. 11.4). Therefore, healthy aquatic ecosystems sustained these pool sites whereas riffle accumulation led to the riparian habitat expansion toward the bank margin. Based on the above view, habitat quality score is given with the following of Calderon and An 2016; Yang et al. 2017, 2018; Bhattacharya et al. 2019a, b (Table 11.3).

3.1.4 Thalweg Shifting

Thalweg line means a line joining the channel's deepest points along the longitudinal dimension, but many anthropogenic activities like sand mining make artificial pits pools on the river bed near the thalweg line. Subsequently, the land strip in the channel becomes divided along the thalweg line, thus the entire off-channel pits are directly converted into in-channel pits (Rinaldi et al. 2005; Ghosh et al. 2016). Therefore, thalweg shifting has highly dominated mined rivers (Bhattacharya et al. 2019b). In this alluvial study reach, maximum thalweg shifting has occurred across the CD (102 m) and EF (80 m) section than AB (39 m) section (Fig. 11.4).

Table 11.3 Impacts of channel geomorphic response parameters on riverine habitat system and score assignment

Condition category	Channel geomorphic responses impact on hydroecological status	Score	Observation site
Pool-riffle alteration (Calderon and An 2016; Yang et al. 2017, 2018; Bhattacharya et al. 2019a, b)			
Optimal	Large shallow with small deep pools are present; riffle is absent	4	Pit sites
Suboptimal	Most of the pools are large-deep; very few shallow; some pools are converted into the riffle	3	Mining/pit sites
Marginal	Shallow pools large expanded than a deep pool; new riffle enormously deposited	2	Mining sites
Poor	Pools are completely absent; riffles are dominated	1	Sandbar sites
Thalweg shifting (Yang et al. 2017, 2018; Bhattacharya et al. 2019a,b)			
Optimal	Absent of thalweg shifting; stream behave like normal pattern; absent of dredging	4	Sandbar sites
Suboptimal	Some thalweg shifting occurred along the mining pool sites; dredging of channel is present	3	Pit sites
Marginal	Embankments or shoring structures both are present along both banks; thalweg shifting rate is 40–80%	2	Mining sites
Poor	Massive altered or degraded instream habitat; bank shore is more dominated; shifting rate of >80%	1	Mining/pit sites
sinuosity index (Yang et al. 2017, 2018; Bhattacharya et al. 2019a,b)			
Optimal	Meandering bend has increased the stream length 3–4 times longer in respect of a straight line	4	Sandbar sites
Suboptimal	Bend increased at the rate of 2–3 times longer than a straight channel	3	Sandbar sites
Marginal	Bend increased at 1–2 times longer than a straight channel	2	Mining sites
Poor	Straight channel flow; mining-induced channelization able to straight waterways for a long distance	1	Pit sites
Water flow (Yang et al. 2013, 2017, 2018; Yu et al. 2019)			
Optimal	Both lower banks are filled up by water; low amount of channel substrate is exposed by mining	4	Sandbar sites
Suboptimal	>75% channel filled up by water; channel substrate is exposed of <25% by sand mining	3	Mining sites
Marginal	25–75% channel filled up by water; maximum riffle substrates are exposing by sand mining	2	Mining sites
Poor	Scarcity of water for filling the lower river bank; huge exposing creates standing pools in mining sites	1	Pit sites

In Kangsabati River, maximum healthy aquatic habitat reaches near the high shifted thalweg channel (Fig. 11.5). Thalweg shifting is directly integrated with the aquatic habitat system, meanwhile wetted mining pit initiated new aquatic habitat either on the channel bed or riparian margin with the supply of nutrient mixed suspended sediments and healthy PP such as BOD, DO, and $_P{}^H$ (Dutta et al. 2017). In this

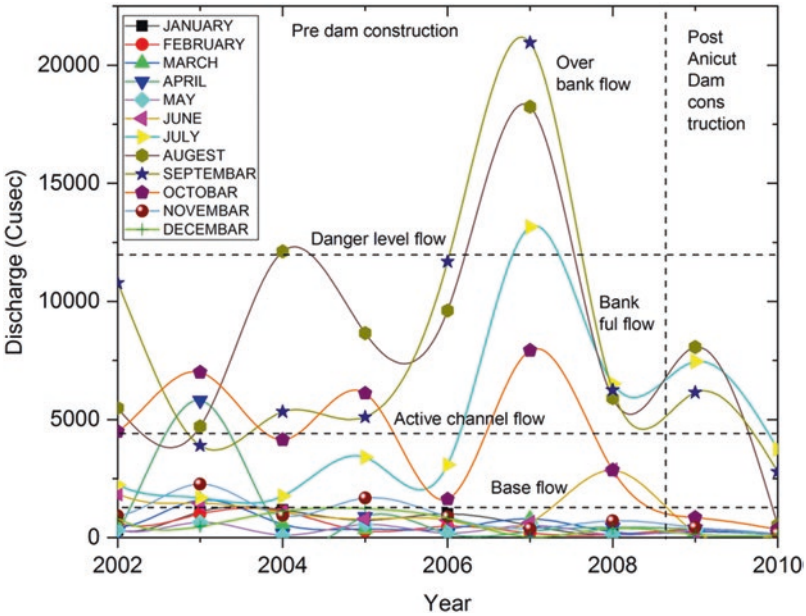

Fig. 11.5 Flow discharge in different months at Mohanpur station (2002–2010)

respect, the score is assigned with the following of Yang et al. 2017, 2018; Bhattacharya et al. 2019a, b (Table 11.3).

3.1.5 Channel Planform Change

Channel morphology is entirely dependent on planform change caused by interruption of fluvial and sediment hydraulics assessing by sinuosity index (P), braiding index (Bi), and braiding channel ratio (B) (Friend and Sinha 1993; Khan et al. 2018). Channel planform controlled the meandering bend for aquatic habitat diversity, longitudinal connectivity for riparian, and bank habitat diversity in accordance to interlinking of stream energy, biotic exchange, and sediment transition with river morphology (Yang et al. 2018). In this study, the P value gradually reduces from 1.68 (2002) to 1.58 (2016), but the B value increases from 1.44 (2002) to 1.18 (2016). In addition, the Bi value decreases from 1.64 (2002) to 1.61 (2016) in the entire study reach (Fig. 11.4). This result demonstrated that sand mining induced planform change interrupted the three-tier habitat systems in the channel. On the other hand, the sinuosity index is considered for assigning the habitat status with the following of Yang et al. 2017, 2018; Bhattacharya et al. 2019a,b (Table 11.3).

3.1.6 Water Flow Regime

Flow regime plays a crucial role to preserve the basic ecological structure and its function of the riverine ecosystem through the preserving of key hydroecological processes in dominant species community, that is, sediment transition, biotic exchange, biotic spawns, energy, and pollution dilution (Song et al. 2007; Yang et al. 2013, 2018). Ecological richness in the river ecosystem is fully maintained by the flow regime along with its influences on the composition of flora and fauna including fish and macroinvertebrates (Dutta et al. 2017). Nutrient accumulation abounded the bank margin and then trapped in pits. Seed distribution and emigration of aquatic species are highly proceeds during bankfull flow while aquatic species are well established during base flow regime due to the absence of maximum turbidity and least sediment transition. In this study, the Anicut dam (after 2009) plays a big role to control the flow regime along the lower course by maintaining the four major principles (Fig. 11.5). Active channel flow (principle 1) generated various channel geometry for habitat complexity that helped to grow up the biotic diversity in riparian microhabitat patch during July and October, whereas bankfull flow (principle 3) triggered the lateral connectivity along the longitudinal course as well as supplied nutrients over the floodplain and bank margin microhabitat patch during August and September. On the other hand, base flow (principle 2) always maintains the life history patterns like spawning during the winter lean period (November–January) and summer lean period (March–June), but the natural regime (principle 4) always discourages the invasions process during extreme peak flow (Table 11.3). Therefore, it is said that the flow regime intensively signified the river habitat system.

3.2 Analysis of Ecological Indicators

Spatial extent of riverine land cover and land use patterns, microhabitat patch area, biodiversity indexes, water quality index is considered for detecting the threshold range of ecological indicators in floodplain, riparian and aquatic ecosystems. Based on the rank score, ecological indicators comprehensively reflect the hydroecological succession in the entire study reach.

3.2.1 Riverine LULC

Land use pattern is considered as leading primary factor for playing of two important roles, that is, declining the aquatic biodiversity in river bed ecosystem (Dutta et al. 2017) and modification of riparian and floodplain micro habitat patch across the channel (Lorenz and Feld 2013; Verdonschot et al. 2016). Several hydroecological attributes including PPs, that is, DO, TDS, $_{P}^{H}$, salinity, BOD, and Mg^{2+} in different sites, that is, sandbar, mining, and pits are greatly variable caused by different land use patterns that have a significant responsibility to variable species distribution as well as species diversity (De Silva et al. 2007; Dutta et al. 2017). Eight LULC patterns named as the agricultural field, brick field, channel, mining sites, riparian vegetation, sandbar, sparse vegetation, and healthy vegetation were taken to analyze the riverine land cover dynamics in the Kangsabati River in the year between 2002 and 2016, respectively (Fig. 11.6a, b). Excessive sand mining creates many pits along the channel bed in mining sites, thus this land cover is considered in LULC during 2016. Brick field (3.76%), sparse vegetation (6.07%), and mining pit (13.14%) gradually increased, but sandbar (−15.05%), dense vegetation (−4.66%), mining sites (−2.23%), and channel (−1.05%) drastically decreased in the last 14 years throughout the study area (Table 11.4). LULC result depicted that massive sand mining and Anicut dam construction both exaggerated the inverse riverine land cover pattern that causes the degradation of the riverine habitat ecosystem.

3.2.2 Microhabitat Patch

Microhabitat patch represents the meso-type habitat system, which involved the structuring of micro invertebrate assemblages in a stream (Beisel et al. 1998; Ralf et al. 2015), fragmented riparian and native species vegetation patch caused by human activities (Dutta et al. 2017; Yang et al. 2018). Figure 11.6a, b reveals that native agricultural crops, scatter and dense June patch are settled along the bank margin, whereas grass colony, riparian vegetation lives across the riparian and floodplain sites, but healthy aquatic vegetation, healthy channel vegetation, fragile aquatic and vegetation in sandbar are assembled along the channel over the Kangsabati River (Fig. 11.7a,b). Microhabitat patch in this study reach demonstrated that scattering June patch (18.98%), healthy channel vegetation (13.49%), aquatic vegetation (2.22%), and fragmented grass colony (1.46%) have increased at an alarming rate while sandbar dominated vegetation (−8.37%), native agricultural crops (−3.15%), dense June patch (−0.48%) have gradually decreased during 2002–2016 (Table 11.5). Therefore, it can be said that human-induced riverine land cover change triggered the species' composition, richness, ecosystem function in microhabitat over the channel. Subsequently, habitat quality score is given with the following of Mosner et al. (2015), Lorenz and Feld (2013), Bhattacharya et al. 2019b (Table 11.8).

3.2.3 Water Quality Index

PP of water quality is considered as a comprehensive attribute for determining the interaction amongst the various ingredients made by physical, chemical, and biological components (Yang et al. 2018; Yu et al. 2019). Consequently, the water quality index is prepared by several PP to detect the water quality status following the five different quality scales, that is, excellent (0–25), good (26–50), poor (51–75), very poor (76–100), and unsuitable (>100) suggested by Ramakrishnaiah et al. (2009), Yadav and Kumar (2011). In terms of PP, maximum TDS (504 mg/L), DO

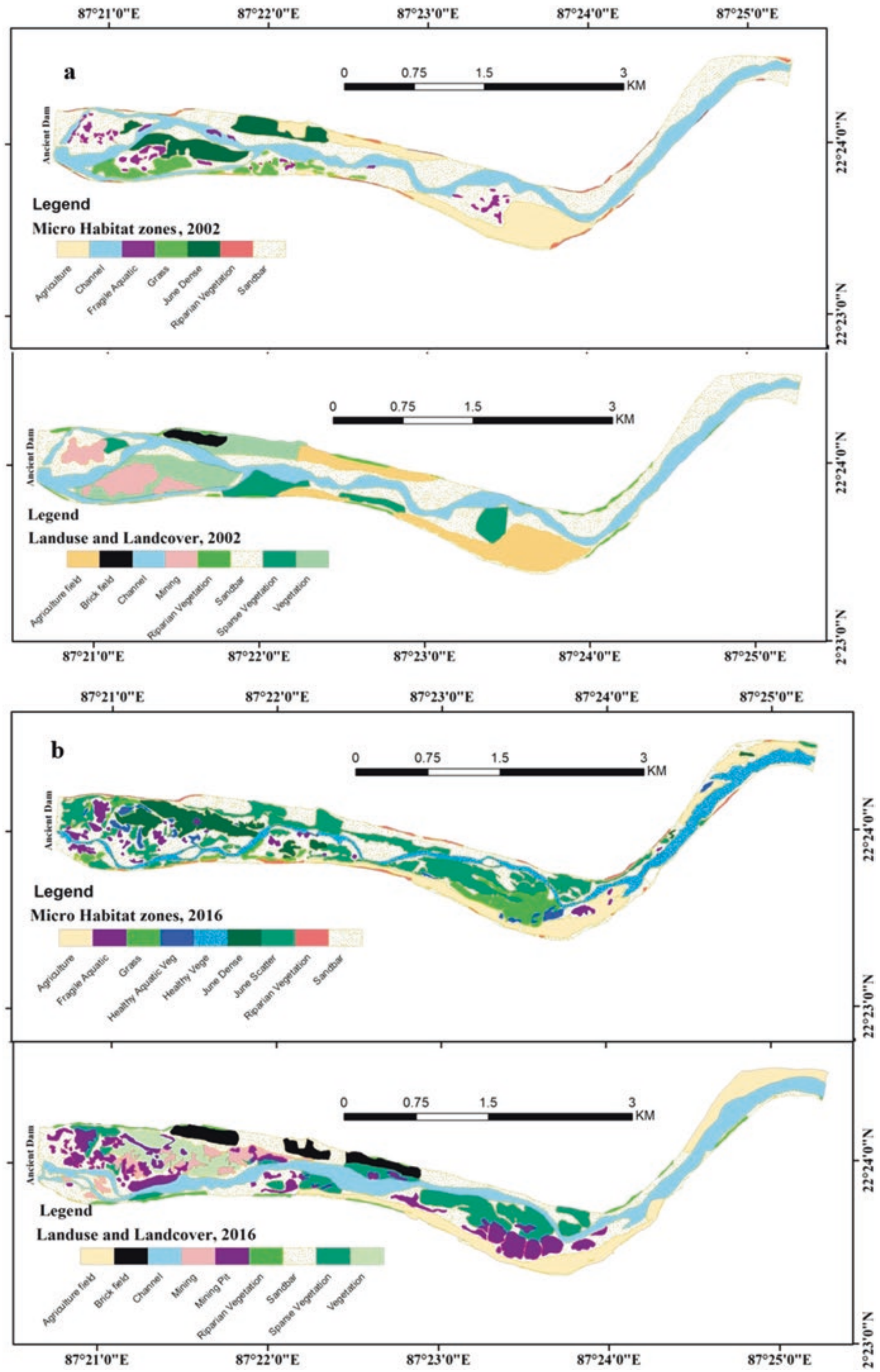

Fig. 11.6 Riverine land use land cover and microhabitat zone: (**a**) 2002, (**b**) 2016

Table 11.4 Riverine land use and land cover in 2002 and 2016

LULC	2002	2016
Agriculture field	0.622342	0.580659
Brick field	0.065047	0.215442
Channel	1.101752	0.97925
Mining	0.28504	0.172381
Riparian vegetation	0.08642	0.076967
Sandbar	1.519742	0.789473
Sparse vegetation	0.287434	0.516961
Vegetation	0.474826	0.248637
Mining pit	0	0.541791

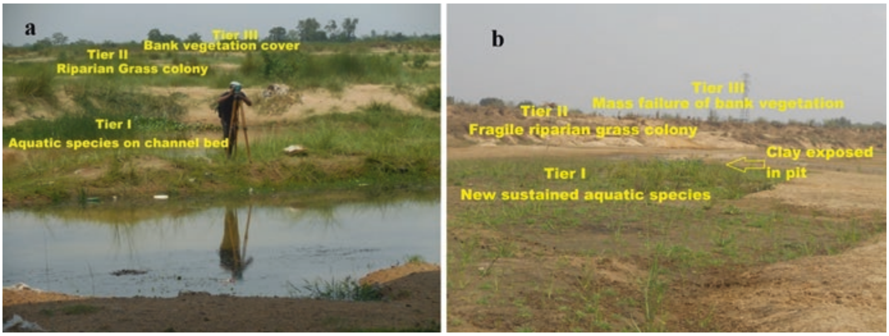

Fig. 11.7 Three tier habitat systems: (**a**) sandbar dominated sites, (**b**) mining dominated sites

Table 11.5 Microhabitat zone in 2002 and 2016

2002		2016	
Name	Area Sq km	Name	Area Sq km
Agriculture	0.580332	Agriculture	0.440116
Channel	1.085378	June scatter	0.844672
Fragile aquatic	0.120748	Fragile aquatic	0.127977
Grass	0.178561	Grass	0.243874
June dense	0.33169	June dense	0.309958
Riparian vegetation	0.078715	Riparian vegetation	0.08598
Sandbar	2.070043	Sandbar	1.69772
		Healthy aquatic vegetation	0.09896
		Healthy vegetation	0.596429

Table 11.6 Water quality parameters in sandbar, mining, and pit sites

Sites	WQI	BOD	Salinity	TDS	Turbidity	Conductivity	PH	DO	Mg
Sandbar	63.7	1.5	0.161	152.35	7.84	256	6.25	0.98	4.56
Sandbar	63.63	1.5	0.235	252.4	7.87	515	6.56	0.77	1.952
Sandbar	55.16	1.4	0.141	147.5	8.36	301	6.566	0.87	3.322
Mining	83.34	1	0.458	554.29	5.65	123	5.86	0.87	6.425
Mining	42.06	1.3	0.172	184.2	8.74	376	7.1	0.98	2.65
Mining	76.92	1.5	0.767	773.6	4.76	156.7	6.392	0.85	6.364
Mining-pit	48.11	1.4	0.597	65.25	5.62	78	6.58	0.69	2.53
Mining-pit	39.81	1.2	0.144	151.4	5.96	310	7.12	0.89	3.227
Mining-pit	24.024	1.5	0.048	46.1	1.79	98.9	7.69	0.67	10.92

(0.9), salinity (0.465 ppt) have been identified in mining sites whereas maximum $_P{}^H$ (7.13) and Mg^{2+} (5.56 ppm) have predominantly concentrated in mining pit sites (Table 11.5). In addition, conductivity (358 µS/cm), BOD (1.47), and turbidity (8.02 NTU) are intensively significant in sandbar sites in the entire reach. On the other hand, WQI in Kangsabati River is observed as good (24.024) to very poor (83.34) category, despite this observation range, the average quality scale always staying in the poor category (55.19) (Table 11.6). Maximum WQI is concentrated in mining sites (67.44) caused by appearing of low BOD, conductivity, and high TDS, DO and salinity, but low-quality index has increased in pit sites (37.31) with the increase of $_P{}^H$, Mg^{2+}, BOD, and decreasing of salinity, conductivity, turbidity over the middle course. Moreover, moderate WQI has been staying in sandbar sites (60.83) due to increases in conductivity and turbidity, as well as decreases in DO, and $_P{}^H$, respectively. PP and WQI status demonstrated that sand mining induced water quality has inversely changed in mining and pit sites than sandbar sites (Table 11.8), thus aquatic ecosystem firmly deteriorated and fragmented over the channel.

3.2.4 Biodiversity Index in Microhabitat Patch

Biodiversity indices are effective measurement tools in microhabitat ecosystems for computing the species abundance, richness, evenness, and functional traits of species (Supriatna 2018; Bera et al. 2020). In the middle course of Kangsabati River, the Simpson index of dominance ranges from 0.29 to 0.15 with an average of 0.194, where the maximum average is found in sandbar sites (0.23) than mining (0.19) and pit sites (0.16) (Table 11.7). Contrastingly, the Shannon-Wiener index of diversity varies from −1.54 to −0.68 with an average value of −1.25 over the reach where dense diversity is concentrated in mining pit sites (−1.51) and mining sites (−1.149) than sandbar sites (−1.11), respectively. Both indices reveal that species diversity in microhabitat patch has been greatly affected by instream and flood plain sand mining, thus this result is similar to that of Bhattacharya et al. (2019c). On the other hand, Marglef index of species richness values ranges from 3.62 to 2.61 covering the average value of 3.06 in this selected reach where high richness is observed in

Table 11.7 Biodiversity indexes in sandbar, mining, and pit sites

Site	Simpson index	Sd	Sr	n	Margalef's index	Diversity index	Pielou index
LS1	0.196078431	0.803922	5.1	18	3.186559081	−1.31	−1.043598099
LS2	0.188948307	0.811052	5.292453	34	2.611854434	−0.68	−0.444015254
LS3	0.198863636	0.801136	5.028571	33	2.634154284	−1.34	−0.882441685
LM1	0.285714286	0.714286	3.5	8	3.321928095	−1.18	−1.306625051
LM2	0.19047619	0.809524	5.25	7	3.549883987	−1.109	−1.312273781
LM3	0.21978022	0.78022	4.55	14	2.617508609	−1.158	−1.010358323
LP1	0.152173913	0.847826	6.571429	24	3.622633876	−1.5	−1.086790163
LP2	0.152148664	0.847851	6.572519	42	3.080241605	−1.54	−0.948714414
LP3	0.168163265	0.831837	5.946602	50	2.94295955	−1.506	−0.886419417

mining pits (3.22) and mining sites (3.16) than sandbar sites (2.81). While Pielou index of species evenness varies from −1.31 to −0.88 maintaining the entire average of −0.991, and maximum species evenness has been found in mining (−1.21) and pit sites (−0.97) than sandbar sites (−0.79), respectively. Average trends of Simpson index of dominance, Shannon-Wiener index of diversity, Marglef index of species richness, and Pielou index of species evenness have been sharply declined in sandbar sites of middle course caused by huge sand extraction than natural replenishment. Moreover, the Anicut dam has an important role to trigger the interruption of biodiversity indices throughout the sites. Based on species diversity, richness, and evenness, habitat quality score is assigned in Table 11.8 with the following of Gothe et al. (2015), Bhattacharya et al. (2019c).

3.3 *River Habitat Assessment*

Based on the CF value of habitat assessment parameters, habitat quality has been classified into four classes such as optimal (>4), suboptimal (4–3), marginal (3–2), and poor (<2), respectively. However, separate habitat parameters are considered to assess the quality from the river bed to the bank margin. Geomorphic responses are only considered for assessing the channel bed habitat quality but geometric, sediment facies and deposition, hydroecological parameters are used to determine the habitat quality in bed, riparian, and bank sites. Quality assessment score denotes that maximum poor health quality is observed in mining affected bed, pit dominated bed, mining affected riparian, pit dominated riparian, and mining affected bank margin (Tables 11.9, 11.10, 11.11). Marginal health quality concentrated in pit affected bank margin while optimal quality found in sandbar sites of riparian and bank margin. Moreover, suboptimal quality is only found in sandbar-dominated river bed sites (Fig. 11.8). Quality score demonstrated human activities like sand mining greatly affected the entire channel than riparian and bank sites.

Table 11.8 Impact ecological parameters on riverine habitat system and score assignment

Condition category	Ecological indicators impact on hydroecological status	Score	Observation site
Vegetation cover (Mosner et al. 2015)			
Optimal	>90% stream bank surface including riparian zones covered by native vegetation; healthy patch	4	Sandbar sites
Suboptimal	70–90% bank surface and riparian zones covered by native vegetation; moderate achievement of healthy vegetation patch	3	Sandbar sites
Marginal	50–70% stream bank surface cover; cropped vegetation and potentiality of plant vegetation affected by mining	2	Mining sites
Poor	<50% stream bank surface covered; degraded patch caused by mining	1	Pit sites
Riparian vegetation (Lorenz and Feld 2013; Bhattacharya et al. 2019b)			
Optimal	Width of the riparian zone as >18 m; human impacts are absent	4	Sandbar sites
Suboptimal	Width of 12–18 m; minimum sand mining present	3	Pit sites
Marginal	Width of 12–18 m; consequences of sand mining abruptly increased	2	Mining sites
Poor	Width of <6 m; riparian vegetation is absent across the bank along the mined river	1	Mining sites
Water quality (Yang et al. 2017, 2018; Bhattacharya et al. 2019c)			
Optimal	High turbidity; DO −1.59 mg/h; BOD-3.1 conductivity 465 s/cm; pH 7.7; total dissolved sediments 228.7 ppm and saline 0.237 psu for healthy aquatic patch	4	Sandbar sites
Suboptimal	Medium turbulent flow and high DO1.07 mg/h; whereas BOD increasing 5.6; so, P 7.8; TDS 198.5 pp. conductivity −477 s/cm. Forms initiating floating grass colony	3	Pit sites
Marginal	Low turbidity; high conductivity-.477 s/cm total dissolved sediments 0.249 ppm; PH-7.8; BOD-3 leads to a mix stable habitat near riparian	2	Mining sites
Poor	No turbidity; pH -8.1; TDS-228.1 ppm; salinity-0.237 psu and BOD-below 3 helps stable riparian habitat	1	Mining sites
Biodiversity index (Gothe et al. 2015; Bhattacharya et al. 2019c)			
Optimal	Maximum species diversity; moderate species richness and low evenness	4	Sandbar sites
Suboptimal	Moderate species diversity; maximum species richness and medium evenness	3	Pit sites
Marginal	Maximum species diversity and moderate species richness; medium species evenness	2	Mining sites
Poor	Moderate species diversity; low species richness; high species evenness	1	Mining sites

Table 11.9 Quality assessment of habitat parameters along the channel bed

Habitat parameters	Sandbar bed	Mining bed	Pit bed
Geometric attributes			
Channel width	4	2	1
Channel depth	3	1	1
Sedimentary facies			
Sediment facies	4	3	1
Sediment deposition	4	2	1
Geomorphic responses			
Pool-riffle sequence	1	2	4
Channel meandering	4	2	1
Thalweg shifting	4	2	3
Flow velocity	4	3	1
Hydroecological attributes			
Vegetation patch	4	1	2
Water quality	4	1	3
Biodiversity index	4	1	3
Quality score	3.33	1.67	1.75
River habitat quality	Suboptimal	Poor	Poor

Table 11.10 Quality assessment of habitat parameters along the riparian

Habitat parameters	Sandbar riparian	Mining riparian	Pit riparian
Geometric attributes			
Riparian width	4	2	1
Sedimentary facies			
Sediment facies	4	1	1
Sediment deposition	4	1	2
Hydroecological attributes			
Vegetation cover	4	2	1
Riparian vegetation	4	1	3
Biodiversity index	4	1	3
Quality score	4	1.33	1.83
River habitat quality	Optimal	Poor	Poor

River habitat assessment assessed the key roles of significant parameter likes geometric, sediment, geomorphic responses, and ecological attributes in three-tier habitat system (Yang et al. 2018). Moreover, substrate composition, water depth, flow velocity plays a significant role to determine the microhabitat of the riverine ecosystem (Maddock 1999). In tier I habitat, massive sand mining exposed the substrate of sedimentary facies along the channel bed; thus, loose sands are enormously deposited on the bed due to breaking down of compactness in sediment. In addition,

Table 11.11 Quality assessment of habitat parameters along the bank margin

Habitat parameters	Sandbar of bank	Mining of bank	Pit of bank
Geometric attributes			
Bank stability	4	1	3
Sedimentary facies			
Sediment facies	4	2	1
Sediment deposition	4	1	2
Hydroecological attributes			
Vegetation cover	4	1	2
Biodiversity index	4	1	3
Quality score	5	1.2	2.2
River habitat quality	Optimal	Poor	Marginal

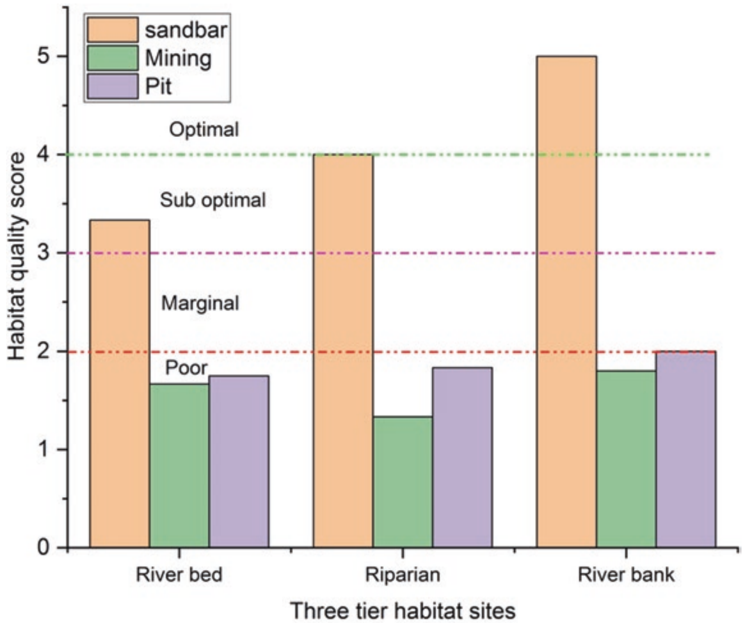

Fig. 11.8 Habitat quality score in sandbar, mining, and pit sites of three-tier habitat system

mining induced turbulent flow increases the pool-riffle alteration and thalweg shifting; subsequently, channel depth continuously increases while channel width and meandering both gradually reduce over the courses.

Contrastingly, high conductivity, turbidity, and salinity deteriorated the water quality in mined beds, as a result, species richness reduces and species evenness increases. Therefore, unstable substrate, inverse geomorphic responses, and deteriorated water quality are triggering the habitat quality to reach a poor category in the entire mined channel. On the other hand, sandbar-dominated channel bed increases the channel width and reduces the depth and inverse geomorphic responses; thus, healthy aquatic ecosystems maintained the suboptimal habitat quality through the offering of stable substrate cover, sediment compactness, and good water quality. In tier II habitat, floodplain sand mining reduces the riparian width along with exposing the clay layer; as a result, mixing of sand, silt, and clay hampered the vegetation patch. On the other hand, huge alluvial deposition gradually increases the width of riparian sites; as a result, healthy grass colonies represented the optimal habitat quality through the generating of vegetation succession. Biodiversity index revealed that maximum species richness dominated in sandy riparian sites with the presence of stable sedimentary facies and alluvial fans. While species evenness is increased in cliff sites caused by the massive down cutting of facies for floodplain mining. In tier III habitat, instream and floodplain sand mining both increased the bank erosion and shifting rate along the bank margin, as a result, vegetation cover is degraded caused by overgrowing bank instability and mixing of sand, silt, and clay in sedimentary facies. Contrastingly, sediment accumulation reduces the bank erosion and shifting rate; thus, vegetation cover reaches optimal habitat quality category resulting from bank stability and widen finer clay layer. Moreover, other anthropogenic activities like channelization, dam construction, and bridge crossing have hampered the riverine habitat ecosystem.

4 Conclusion

Assessment of river habitat system provided greater information about the potential causes of human-induced impairment in particularly sand mining using habitat quality score. In respect of quality score, maximum optimal and suboptimal quality categories were found in sandbar dominated riparian, bank, and bed habitat sites, while poor and marginal quality category concentrated in mining affected bed, riparian, and bank margin sites, respectively. Several geometric and planform techniques demonstrated that sand mining greatly affected the channel depth, width, riparian margin, and bank stability through the occurring of pool-riffle alteration, thalweg shifting, and turbulence water flow. Sedimentary facies analyzes depicted that sand mining changes the compactness of sediment and textural attributes across the bed to the bank, at the same time micro habitat patch faces more disturbance in mining and pit sites than sandbar sites. On the other hand, the water quality index revealed that maximum mining affected physiochemical parameters of water in channel fall under poor category but good water quality observed in nonmining or sandbar site. Biodiversity index showed that species diversity and richness are increases in healthy aquatic ecosystems, riparian grass colony and bank vegetation cover of sandbar dominated sites, while species evenness and fragmented vegetation patch observed in mining affected channel, riparian, and bank margin. Over extraction of sand mining changes, the channel geometry, sedimentary facies, sediment deposition, and microhabitat zone resulting from negative riverine land use land cover change caused by generating of inverse geomorphic responses in a mined channel. Therefore, water quality has deteriorated in the entire channel, subsequently, species diversity and richness have been decreasing but species evenness has gradually expanded in three-tier habitat systems.

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Chapter 12
Physicochemical and Microbial Indicators for Water Quality Assessment in an Industrial Catchment of River Damodar, India

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Abstract Surface water quality of river Damodar in an industrial catchment was assessed in two different seasons (pre- and post-monsoon). Eleven discharge points were selected for analysis of physical, chemical, and microbial variation between sampling sites for two seasons. Physicochemical analysis showed a very high concentration of each parameter in sample water and exceeded its permissible limit for drinking in the post-monsoon season. Untreated sewage and wastewater from nearby industries supplied pathogenic matters to the riverbed. Microbial indicators such as Total Viable Count (TVC), Total Coliform (TC), and Faecal Coliform (FC) indicated high positive correlation with physicochemical parameters such as temperature, Total Hardness (TH), cadmium, iron, Biological Oxygen Demand (BOD),

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oil and grease in river water. Highest TVC and coliform bacteria were observed in the S8 sample near Durgapur thermal power plant. Discharge of hot wastewater from a power plant is the principal cause of higher microbial growth at this site. Pollution Index (PI) was analysed for physicochemical quality of river water and this analysis showed the highest value of PI in a S8 sample during both seasons. Hypothesis testing by 't' test indicated there was a significant variation ($\rho > 0.01,0.05$) in the mean value of all parameters in two seasons and rejected null hypothesis. Mean difference of PI values (t = 20.59) also indicated a high difference between post- and pre-monsoon seasons. 'Lockdown' period of COVID -19 pandemic helped to improve river water quality due to the closing of all industries. Very low discharge of wastewater to the riverbed noticeably improved water quality during the pre-monsoon season. After 'unlock' of economic sectors proper management and mitigation of river pollution should be very necessary to sustain environmental quality and protect human health from pathogenic hazards in this study area.

Keywords Environmental quality · Lockdown · Microbial indicators · Pollution index · Proper management · Water quality

1 Introduction

Freshwater plays a very important role to sustain life and growth of socioeconomic environment, worldwide. Rapid urbanisation and industrial development put daily its adverse impact on the environment and thus adequate availability of clean and fresh water become a major challenge to the globe. Surface water like rivers, lakes, channels, and so on, serves near about one-third of water for drinking purposes in the whole world (Edokpayi et al. 2018). Surface water bodies are also used as discharge ground of industrial, agricultural and domestic wastes (Tukura et al. 2009). This led to pathogenic contamination of water and poses a terrible threat to human health risks (Srivastava et al. 2017). According to World Health Organisation (WHO), annually 525,000 death cases of children with diarrhoea disease were reported due to the intake of untreated drinking water in the world (WHO 2018). In India, waterborne disease affects approximately 37.7 million people in every year and 15 million children are lead to death by diarrhoea (Khurana and Sen 2008). Therefore, regular monitoring of water quality should be very necessary to protect human health and the aquatic environment.

Water quality is constituted by three properties as physical, chemical and biological. Physical parameters are related to temperature, pH, turbidity, conductivity, and so on, and chemical parameters are related to ion concentration, metallic and toxic elements contamination to the water. Biological parameters such as total coliform (TC) and faecal coliform (FC) are the most important indicators of pathogenic contamination of water (APHA 1995). Coliform bacteria are facultative anaerobic or aerobic, rod-shaped, gram-negative, motail or non motail groups of microbes (Li and Liu 2018). Coliform bacteria present in drinking water are a strong indicator of faecal contamination. Total Coliform (TC) denotes the complete bacterial count of coliform, a subset of TC is termed as Faecal Coliform (FC) and a fraction of FC is called E. coli bacteria (Brackett et al. 1993). These bacteria cause serious health hazard to the human body. Biological activities are directly influenced by the physical and chemical stats of the water bodies.

Many researchers have conducted studies for physicochemical and microbial assessment of riverine water quality in various parts of the world by using different water quality analysis techniques (Leong et al. 2018; Haque et al. 2019; Rawal et al. 2018; Kumar et al. 2012; Edokpayi et al. 2018). Damodar river of India originated from the Chotanagpur plateau and flows through mineral-rich coal seam basin. Therefore, heavy industrialisation and urbanisation, and the discharge of numerous harmful wastes to the river water from industries and urban sectors of the riverbank are deteriorating the water quality of the Damodar river. Effluents of waste materials and toxic liquids are flowed by various nallas from different industries and coal mining fields and contribute adverse impact on the water environment. Water of this river is largely used for drinking, agriculture, recreational and industrial purposes in its catchment area. Some studies on the river Damodar were conducted previously for assessment of arsenic load (Acharya and Shah 2007), chironomids development (Bhattacharya et al. 2005), aquatic worms (Pandit et al. 1996), industrial impact (George et al. 2010; Tiwary 2004), physicochemical and microbial analysis (Chatterjee et al. 2010) in different times. But the seasonal variation of physical, chemical and microbial quality on river Damodar in the highly industrialised and urbanised catchment has not been analysed before. Thus, the objective of our study is to assess the seasonal variation of physicochemical and microbial quality of water in different discharge sites of industrial waste in the riverbed and thereby provide possible management strategies to the practitioners to check the water pollution.

2 Study Area

The present study area is situated between Panchet dam (23°40′50´´N/86°44′49″E) and Durgapur barrage (23°40′50´´N/87°18′39″E) with a length of 65.37 km long river bed (Fig. 12.1). Total eleven (11) discharge points have been identified in the riverbed to collect samples for analysis of the river water quality. Location of the sample points is listed in Table 12.1.

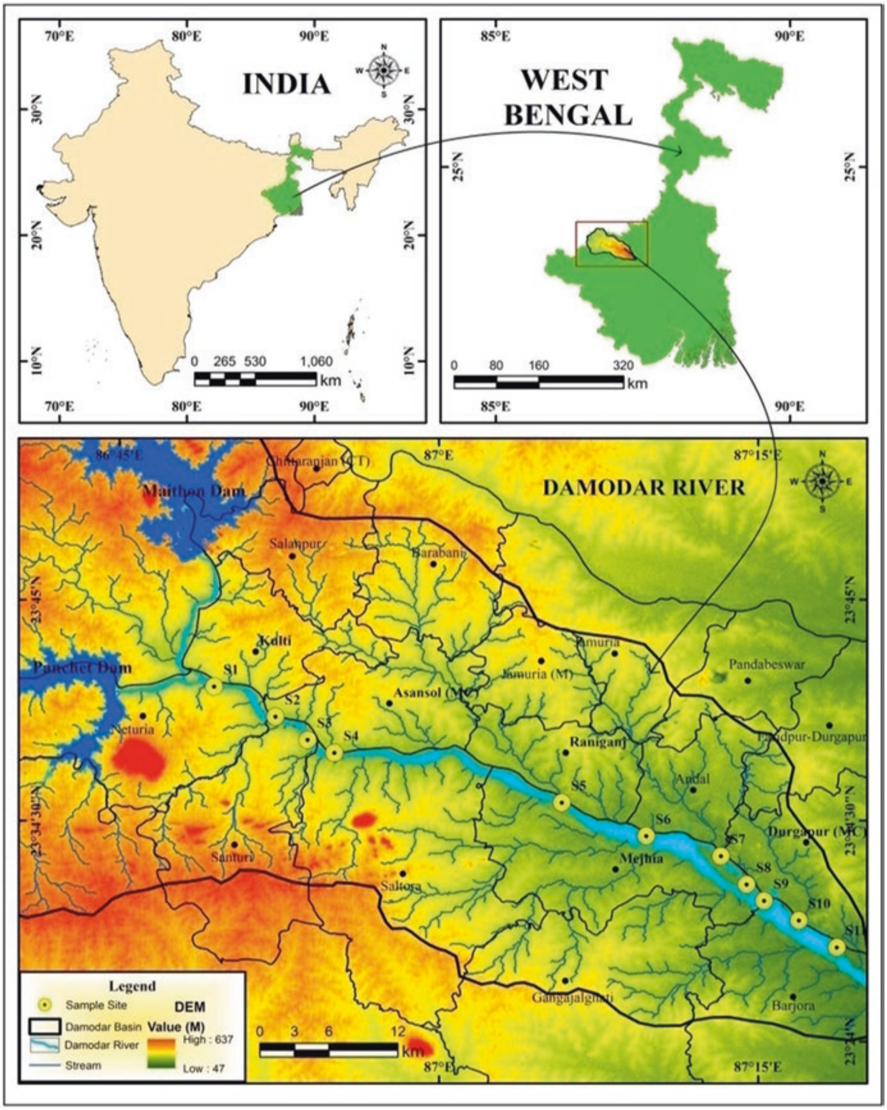

Fig. 12.1 Location map and sampling point of river Damodar

3 Materials and Methods

3.1 Collection of Samples and Physicochemical Analysis

Samples were collected in two different seasons as post-monsoon (December 2019) and pre-monsoon (May 2020) season. Half litter of pre-sterilized bottles were used for sample collection and washed with sample water three to four times before

Table 12.1 Geographical location of 11 sample sites on Damodar river bed

Latitude (N)	Longitude (E)	Sample ID	Effluents from nearby industries
23°40′52.5″N	86°49′26.8″E	S1	Perbelia coal mining field
23°38′31.7″N	86°53′38.8″E	S2	Madhukunda sponge iron factory
23°38′11.9″N	86°53′44.9″E	S3	ACC cement factory, Madhukunda
23°37′43.0″N	86°54′55.9″E	S4	Burnpur IISCO
23°35′21.6″N	87°05′47.1″E	S5	Sabitri sponge iron plant, Raniganj
23°32′58.9″N	87°10′53.8″E	S6	Mejia thermal power plant
23°32′49.2″N	87°13′15.3″E	S7	Birla cement corporation, Waria
23°31′28.0″N	87°14′27.9″E	S8	Durgapur thermal power plant
23°30′41.9″N	87°15′17.1″E	S9	Alloy steel plant, Durgapur
23°29′45.2″N	87°16′55.1″E	S10	Durgapur chemicals limited
23°28′28.7″N	87°18′42.4″E	S11	Durgapur barrage

containing and preserved properly for laboratory analysis within 24 h. Samples from five points were mixed and the composite sample was used for laboratory analysis. All the selected parameters were estimated using the standard method as described by APHA (1998). pH, EC and temperature were measured in situ by their portable meters (Hanna HI9811–5; thermo probe model TL1-A). Concentration of TH, Cd, Cr, Fe, NO_3^-, BOD, DO, oil and grease were analysed in the laboratory.

3.2 *Pollution Index (PI) for Drinking Water*

River water suitability to drinking purposes has been analysed by the pollution index (PI) method with minor modifications. This method was first applied by Subba Rao (2012) for the groundwater suitability to drinking and it is a very effective tool to delineate the relative influence of each physiochemical parameter on the overall quality of water. Five steps were followed to calculate PI. At first, each parameter was assigned its relative weight (Rw) from 1 to 5 according to its severity to human health (Table 12.2). In the second step, the weight parameter (Wp) was calculated as dividing Rw of each parameter by summation of all relative weights to assess the relative share of each component to the drinking water quality (Eq. 12.1). In third step, status of concentration (Sc) was computed by dividing the concentration of each parameter in every sample by its standard limit for drinking purpose (Eq. 12.2). Fourth step is involved with overall water quality (Ow) and it was calculated by multiplication of Sc with Wp (Eq. 12.3). Last of all, the pollution index (PI) of water (Eq. 12.4) was estimated by the addition of all Ow values of each parameter (Subba Rao et al. 2018)

$$Wp = \frac{R_W}{\sum R_w} \tag{12.1}$$

Table 12.2 Relative weight (Rw) and Weight parameter (Wp) of physicochemical parameters

Parameters	Standard value	References	Weight (wi)	Relative weight (Wi)
Temperature	30 °C	BIS (2012)	2	0.05
pH	7.5	BIS (2012)	5	0.125
TDS (mg/l)	500 mg/L	WHO (2011)	5	0.125
EC (μS/cm)	730 mg/L	BIS (2012)	2	0.05
TH (mg/l)	200 mg/L	BIS (2012)	2	0.05
Fe (mg/l)	3 mg/L	BIS (2012)	3	0.075
Cd (mg/l)	0.003 mg/L	WHO (2011)	3	0.075
Cr (mg/l)	0.05 mg/L	WHO (2011)	3	0.075
NO_3^- (mg/L)	45 mg/L	BIS (2012)	4	0.1
BOD (mg/L)	5 mg/L	UN EPA (1999)	5	0.125
DO (mg/L)	6 mg/L	ICMR (1975)	5	0.125
Oil and grease (mg/L)	0.5 mg/L	BIS (2012)	1	0.025

$$Sc = \frac{C}{D_s} \tag{12.2}$$

$$O_W = W_p \times S_c \tag{12.3}$$

$$\mathrm{PI} = \sum O_W \tag{12.4}$$

The overall values of PI in pre-monsoon and post-monsoon season have been classified in three groups as insignificant pollution (<1), low pollution (1–1.5) and high pollution (<1.5) of the river water (after modified Subba Rao et al. 2018).

3.3 *Microbial Analysis*

3.3.1 Total Viable Count (TVC)

TVC was estimated by using aliquots of water samples (0.1 ml). Normal saline 0.85% sterile was used for dilution of sample water in 1:9 ratios and examined on nutrient agar media (Hi Media) plates and incubated for 24 h at 25 °C temperature. The bacterial colonies were estimated by plate count number multiplied by the dilution factor (Leong et al. 2018).

3.3.2 Coliform Count (TC and FC)

Most probable number (MPN) method was used for estimation of indicator bacteria as total coliform and faecal coliform (MPN/100) in the laboratory. Three sets of fermentation tubes containing lauryl tryptose broth (Hi Media, India) were prepared for inoculation of 10 mL, 1 mL and 0.1 mL of sample water. Each set was made with a group of five tubes. Incubation period of bacterial growth was 48 h at 37 °C. Positive tubes were identified by gas and acid production in Durham tubes (Srivastava et al. 2017).

Confirmation test was prepared with brilliant green lactose bile (BGLB) broth tubes where a loop full of culture from each positive tube was inoculated and incubated for 24 to 48 h at 37 °C and 44.5 °C for total coliform and faecal coliform, respectively.

Completed test of coliform bacteria was done by a loop full of broth from positive BGLB tubes transferred on an eosin methylene blue (EMB) agar and incubated for 24 to 48 h at 37 °C. Confirmation of complete test was indicated by the formation of colonies with a green metallic sheen.

3.4 Multivariate Analysis

Multivariate analysis such as hierarchical cluster analysis (HCA) is such a useful statistical tool to classify variables into different groups based on their similarity (Zhang et al. 2012). SPSS16 software has been used to generate HCA to identify relations among various water quality parameters to each other for pre-monsoon and post-monsoon seasons.

4 Results

4.1 Hydro-Chemistry of River Water

Statistical summary (mean, SD, kurtosis, skewness, max, min, cv) of the physicochemical and microbial parameters has been presented for post-monsoon (Table 12.3) and pre-monsoon seasons (Table 12.4). Laboratory analysis shows that higher temperature was observed in S5, S8, and S9 samples during the post-monsoon season. Hot wastewater from sponge iron factories, steel plants and chemical industries is directly discharged into riverbeds and thus the temperature of river water increases even in the winter season. Highest level of pH and NO_3^- was found in S7. A waste effluent from the cement industry has noticeably increased pH at this sample point. Highest TDS, EC, TH, Cd were found in the S9 sample. At this location, huge amount of chemically polluted wastewater has been discharged from the nearby steel plants, regularly. Concentration of iron and oil and grease was recorded highest at sampling site S4. Wastewater mixed with numerous ferrous contain from Burnpur IISCO discharged at this site and increased iron in the water. Concentration of Cr was found at its highest level at S10 sample site. Poisonous chemical effluents released with wastewater from the nearest chemical industry contained a high level of Cr with it. BOD was found highest in the S8 sample indicates high mixing of hot

Table 12.3 Statistical summary of parameters in post-monsoon (December 2019) season

Variables	Temperature (°C)	PH	TDS (mg/l)	EC (µS/cm)	TH (mg/l)	Fe (mg/l)	Cd (mg/l)	Cr (mg/l)	No3- (mg/l)	BOD (mg/l)	DO (mg/l)	Oil and grease (mg/l)	TVC (cfu mL^{-1})	TC (MPN/100 mL^{-1})	FC (MPN/100 mL^{-1})
Mean	29.45	7.46	740.65	1157.27	470.91	4.55	0.01	0.11	62.31	12.55	4.87	4.16	23545.45	9263.64	2623.64
Standard deviation	2.21	0.38	50.24	78.50	80.18	0.91	0.00	0.02	15.38	3.47	0.50	1.47	4156.05	1144.79	889.00
Kurtosis	−1.45	−0.34	−1.68	−1.68	−1.29	−1.12	−0.96	−1.31	−1.78	−0.71	−1.36	0.42	−0.56	−0.61	−1.27
Skewness	−0.23	0.75	0.00	0.00	−0.05	−0.48	−0.24	−0.23	0.30	−0.04	−0.14	−1.12	0.25	−0.48	0.63
Range	6.00	1.17	140.80	220.00	230.00	2.64	0.01	0.05	41.50	11.00	1.47	4.57	13,000.00	3600.00	2460.00
Minimum	26.00	7.04	665.60	1040.00	350.00	3.10	0.01	0.08	43.20	7.00	4.20	1.30	17,000.00	7400.00	1560.00
Maximum	32.00	8.21	806.40	1260.00	580.00	5.74	0.01	0.13	84.70	18.00	5.67	5.87	30,000.00	11,000.00	4020.00
CV	7.4943	5.06	6.78	6.78	17.02	20.09	21.65	15.91	24.69	27.7	10.205	35.33	17.651165	12.357896	33.88436

Table 12.4 Statistical summary of parameters in pre-monsoon (May 2020) season

Variables	Temperature (°C)	PH	TDS (mg/L)	EC (μS/cm)	TH (mg/L)	Fe (mg/L)	Cd (mg/L)	Cr (mg/L)	No3- (mg/L)	BOD (mg/L)	DO (mg/L)	Oil and grease (mg/L)	TVC (cfu mL^{-1})	TC (MPN/100 mL^{-1})	FC (MPN/100 mL^{-1})
Mean	27.45	6.92	524.86	820.09	282.00	3.59	0.00	0.02	37.46	7.27	7.31	1.54	13363.64	4341.82	944.27
Standard deviation	2.30	0.48	26.30	41.09	30.38	0.46	0.00	0.02	5.48	2.41	0.60	0.76	5454.96	782.53	134.24
Kurtosis	−1.35	−0.71	−0.67	−0.67	−0.07	−1.91	0.01	0.65	0.91	2.31	0.53	−1.82	−0.69	0.21	−0.56
Skewness	−0.56	−0.16	−0.43	−0.43	0.34	0.32	0.89	1.11	−0.98	1.54	−0.74	−0.11	0.59	−0.55	0.17
Range	6.00	1.60	83.20	130.00	100.00	1.10	0.00	0.05	18.40	8.00	2.00	2.05	16200.00	2600.00	440.00
Minimum	24.00	6.12	480.00	750.00	240.00	3.10	0.00	0.01	25.40	5.00	6.15	0.40	6800.00	3000.00	740.00
Maximum	30.00	7.72	563.20	880.00	340.00	4.20	0.01	0.06	43.80	13.00	8.15	2.45	23000.00	5600.00	1180.00
CV	8.36	6.96	5.01	5.01	10.77	12.77	59.33	66.60	14.63	33.17	8.16	49.51	40.82	18.02	14.22

wastewater from power stations develops favourable conditions for biological activities and increased demand for dissolved oxygen in river water. DO was found its highest concentration in S1 where very low mixing of harmful chemicals, hot water and lower contains dissolved solids (organic/inorganic) help to reduce microbial activities and thus increased level of dissolved oxygen at this site.

In the pre-monsoon season, the maximum temperature was recorded in S5 and S8 sample sites. In this season highest TDS and EC were recorded from S5 sample water. Fe, Cd, Cr has recorded their highest concentration in S4, S7 and S10 sample sites, respectively. pH and BOD showed their maximum presence in S10 and S8 sample points, respectively. Highest DO was recorded in S1 water sample in the study area. '*t'* value of all parameters indicates significant changes ($\rho < 0.05$) in pre-monsoon season from post-monsoon and rejects the null hypothesis (Table 12.5).

Correlation analysis depicts a significant high positive correlation of TH, Fe, BOD, oil and grease, TVC and FC with the temperature of river water in the post-monsoon season (Table 12.6). In this season, high positive correlation was found among TVC, TC and FC. Concentration of Fe, Cd, EC, and TDS indicates high positive relation with microbial communities because these parameters help to create favourable environment for the growth of microbes. In the pre-monsoon season, pH, Cd, oil and grease show a high positive correlation with temperature of water (Table 12.7). BOD shows a very high positive correlation with oil and grease, TVC, and FC of the sample water. In both two seasons DO indicates a negative correlation with microbial parameters of the river water.

Table 12.5 Testing of hypothesis by 't' test of both season

Parameters	Calculated value of t	Critical table value at significant level (α)	
		α: 0.01	α = 0.05
Temperature (°C)	6.06	3.16	2.22
pH	11.01	3.16	2.22
TDS (mg/L)	12.34	3.16	2.22
EC (μS/cm)	12.94	3.16	2.22
TH (mg/L)	7.60	3.16	2.22
Fe (mg/L)	4.99	3.16	2.22
Cd (mg/L)	10.36	3.16	2.22
Cr (mg/L)	13.14	3.16	2.22
No_3^- (mg/L)	6.74	3.16	2.22
BOD (mg/L)	7.96	3.16	2.22
DO (mg/L)	−12.08	3.16	2.22
Oil and grease (mg/L)	8.40	3.16	2.22
TVC (*cfu* mL^{-1})	19.22	3.16	2.22
TC (*MPN*/100 mL^{-1})	33.18	3.16	2.22
FC (*MPN*/100 mL^{-1})	7.03	3.16	2.22
PI	20.50	3.16	2.22

Table 12.6 Correlation analysis of parameters in post-monsoon (*significant at 5% level of variance)

Variables	*Temperature*	*PH*	*TDS*	*EC*	*TH*	*Fe*	*Cd*	*Cr*	*No3-*	*BOD*	*DO*	*Oil and grease*	*TVC*	*TC*	*FC*
Temperature	1														
PH	0.23	1.00													
TDS	0.58	0.29	1.00												
EC	0.58	0.29	1.00	1.00											
TH	0.71*	−0.02	0.57	0.57	1.00										
Fe	0.72*	−0.10	0.38	0.38	0.54	1.00									
Cd	0.58	0.34	0.27	0.27	0.33	0.47	1.00								
Cr	0.52	0.29	0.17	0.17	0.49	0.36	0.42	1.00							
No3-	0.56	0.09	0.78*	0.78	0.45	0.41	0.00	0.12	1.00						
BOD	0.86*	0.24	0.43	0.43	0.49	0.66	0.82*	0.50	−0.34	1.00					
DO	0.09	0.05	−0.14	−0.14	−0.21	−0.10	−0.40	−0.51	−0.24	−0.15	1.00				
Oil and grease	0.76*	0.33	0.40	0.40	0.44	0.80*	0.59	0.48	−0.33	0.83*	−0.12	1.00			
TVC	0.78*	0.18	0.37	0.37	0.43	0.60	0.82*	0.50	−0.26	0.98*	−0.24	0.77	1.00		
TC	0.52	0.04	0.05	0.05	0.17	0.45	0.56	0.50	0.03	0.80*	−0.23	0.66	0.86*	1.00	
FC	0.96*	0.33	0.58	0.58	0.70*	0.73*	0.72*	0.56	−0.43	0.92*	−0.07	0.83*	0.85*	0.60	1

Table 12.7 Correlation analysis of parameters in pre-monsoon (*significant at 5% level of variance)

Variables	Temperature	PH	TDS	EC	TH	Fe	Cd	Cr	No3-	BOD	DO	Oil and grease	TVC	TC	FC
Temperature	1.00														
PH	0.76*	1.00													
TDS	0.33	0.47	1.00												
EC	0.33	0.47	1.00	1.00											
TH	0.09	0.15	−0.53	−0.53	1.00										
Fe	0.37	0.61	0.10	0.10	0.05	1.00									
Cd	0.75*	0.58	0.24	0.24	0.10	0.28	1.00								
Cr	0.47	0.47	−0.03	−0.03	0.30	0.41	0.47	1.00							
No3-	0.01	0.31	0.44	0.44	0.28	0.16	0.02	0.02	1.00						
BOD	0.68	0.49	0.27	0.27	0.10	0.06	0.42	0.32	0.05	1.00					
DO	−0.55	−0.34	−0.11	−0.11	0.07	−0.25	−0.11	−0.17	−0.20	−0.75	1.00				
Oil and grease	0.85*	0.73*	0.19	0.19	0.22	0.59	−0.44	0.52	−0.01	0.72*	−0.75	1.00			
TVC	0.59	0.29	0.20	0.20	−0.23	0.00	0.29	0.43	0.04	0.80*	−0.75	0.61	1.00		
TC	0.39	0.60	0.60	0.60	−0.34	0.20	0.08	0.10	0.58	0.60	−0.59	0.47	0.62	1.00	
FC	0.54	0.32	0.52	0.52	−0.31	0.07	0.10	0.19	0.08	0.80*	−0.80*	0.65	0.83*	0.68	1.00

Table 12.8 Pollution Index (PI) value of sample water in post- and pre-monsoon seasons

Sample No	PI in post-monsoon	Pollution zone	PI value in pre-monsoon	Pollution zone
S1	1.16	Low	0.82	Insignificant
S2	1.53	High	0.89	Insignificant
S3	1.32	Low	0.90	Insignificant
S4	1.76	High	0.99	Insignificant
S5	1.67	High	0.96	Insignificant
S6	1.64	High	0.87	Insignificant
S7	1.70	High	0.92	Insignificant
S8	1.93	High	1.13	Low
S9	1.90	High	1.05	Low
S10	1.81	High	1.10	Low
S11	1.64	High	0.94	Insignificant

4.2 Pollution Index (PI) Analysis

Pollution index of the study area was analysed by following standard values of different parameters set by WHO (2011), BIS (2012), UN EPA (1999) and ICMR (1975) for drinking water quality. The overall values of PI for post and pre-monsoon season is ranged from 0.83 to 1.93 and it has classified as insignificant (<1), low (1–1.5), and high (>1.5) pollution zone (Table 12.8). In post-monsoon and pre-monsoon seasons lowest and highest PI values were found from S1 and S8 sample water, respectively.

4.3 Microbial Analysis

Spatial variation of concentration of microbial bacteria in different sample sites has been presented in Fig. 12.2 for post-monsoon (a) and pre-monsoon (b) seasons.

4.3.1 Total Viable Count

Values of total bacterial counts (cfu mL^{-1}) of 11 sampling points in the post-monsoon season were ranged from 17×10^3 to 30×10^3 cfu mL^{-1} with an average of 23,545.45 ± 4156.05 (Table 12.3). Highest value of TVC was calculated from S8 and S9 sample water in this season. In pre-monsoon, TVC was ranged from 68×10^2 to 23×10^3 cfu mL^{-1} with an average of 13,363.64 ± 5454.96 (Table 12.4). In this season lowest TVC was counted from S1 and highest TVC was counted from S8 sample water. TVC of all samples indicates a very high concentration of bacteria that exceeds the WHO threshold value (100 cfu mL^{-1}) for drinking water in both seasons).

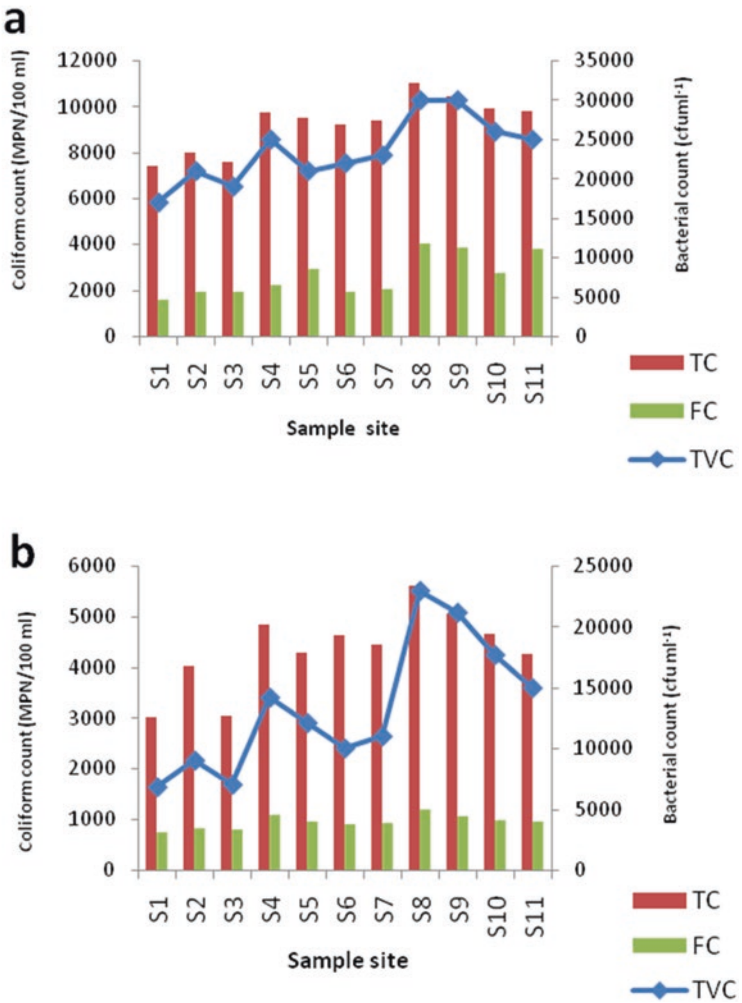

Fig. 12.2 Spatial variation of microbial communities in post-monsoon (**a**) and pre-monsoon season (**b**) at sampling points

4.3.2 Coliform Count

Total coliform count of the post-monsoon season was ranged from 7400 to 11,000 MPN/100 mL $^{-1}$ with an average value of 9263.64 ± 1144.79 (Table 12.3). In this season lowest coliform was recorded from S1 and the highest coliform count was recorded from S8 sample water. In the pre-monsoon season, TC was ranged from 3000 to 5600 MPN/100 mL $^{-1}$ with an average value of 4341.82 ± 782.53 (Table 12.4). Lowest TC was observed from S1 sample and highest TC was observed from the S8 sample. All samples of both seasons exceed the standard limit of TC (3 coli/100 mL) set by WHO for drinking water.

4.3.3 Faecal Coliform Count

Faecal coliform in post-monsoon was ranged from 1560 to 4020 MPN/100 mL with an average of 2623.64 ± 889.00 (Table 12.3). Highest FC was observed from S8 and lowest FC was observed from S1 sample water in this season. In the pre-monsoon season, FC was ranged from 740 to 1180 MPN/100 mL with a mean value of 944.27 ± 134.24 (Table 12.4). No one of the samples indicated its FC values less than WHO standard limit (0 coli/100 mL) for drinking water in both seasons.

4.4 Multivariate Analysis

Hierarchical cluster analysis is a powerful technique of multivariate analysis, which creates clusters among variables or cases (samples) with high similarities within the groups and high dissimilarities between the groups (Sarbu and Pop 2005) by generating Dendogram. In the present study cluster analysis of water quality parameters by HCA has been prepared for post-monsoon and pre-monsoon seasons by using SPSS 16 software. Between group linkages method had selected for clustering and the linkage among parameters was estimated by the squared Euclidean distance method.

Dendogram of post-monsoon (Fig. 12.3) and pre-monsoon (Fig. 12.4) depicts that there is close similarity between physicochemical parameters with coliform bacteria (TC and FC). These variables generated the same cluster or group in two different seasons. TVC is highly related to TC and FC; it creates another group or cluster of coliform bacteria in two seasons.

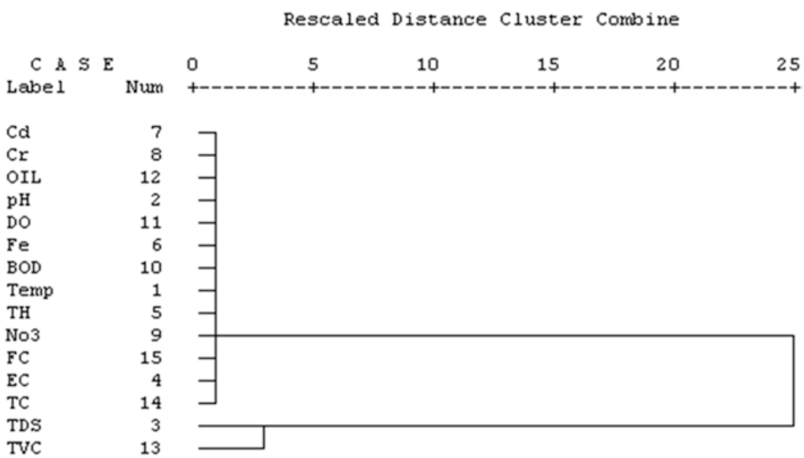

Fig. 12.3 Dendogram of post-monsoon season (dendogram using average linkage (between groups))

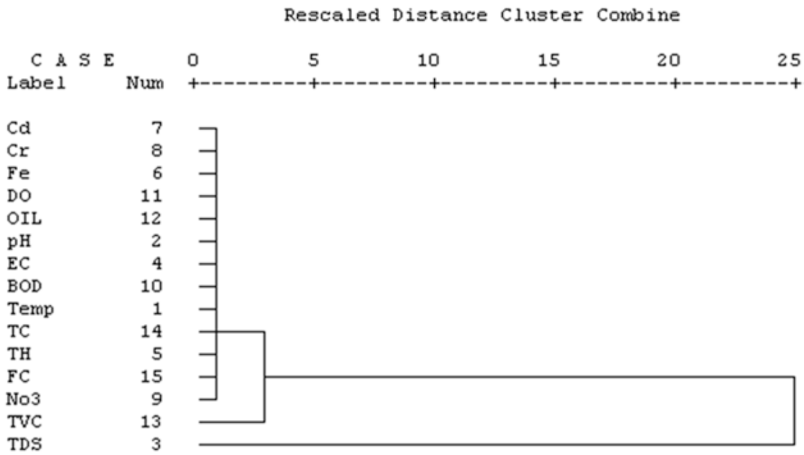

Fig. 12.4 Dendogram of pre-monsoon season (dendogram using average linkage (between groups))

5 Discussions

Physicochemical analysis of eleven (11) sample water shows a huge negative contribution of industrial wastewater to river Damodar in the study area. Samples of post-monsoon season (December 2019) clearly indicate that very high concentration of chemical parameters in different discharge sites of wastewater promotes the growth of microorganisms in riverbeds. Industries like steel plants, sponge iron, thermal power plant (S5, S8, and S9) used cold water for cooling machines and released hot wastewater to the nallas. This hot water was directly discharged to the main river bed and thus the temperature of the river water increased highly. Temperature of surface water is a dominant factor to control pathogenic bacteria activity (Edokpayi et al. 2018). High level of TDS, EC reduces the clarity of water, increases the ability to carry electric current and creates favourable environment for the growth of microorganisms (US EPA 1999). Mixing of chemicals and pathogenic components controls the pH level of river water. In this study area sewage from steel plant (S9) contributes the highest amount of TDS, EC, and TH to the river water. BOD is a very important parameter to indicate biological activities of surface water. Higher activities of microorganisms increase the demand for oxygen in water. Thus, high rate of organic supplements favours the growth of anaerobes in the river ecosystem (Mtui and Nakamurs 2006). DO of surface water indicate biological health and it has an inverse relation with microbes. In this season excessive organic components are being discharged into the river by industries, deplete the level of DO and increased the rate of microbial decomposition in most sample sites (Singh et al. 2002).

Pollution Index (PI) of sample water reveals high pollution load due to high concentration of physicochemical parameters in samples S2, S4, S5, S6, S7, S8, S9, S10 and S11 in the post-monsoon season. Highest TVC was found from S8 because the high temperature of river water helps to create a favourable environment for bacteriological activities. Some previous researches provide information that the bacteriological activities of the river Damodar are highly related to the physicochemical parameters of river water. High concentration of pH, TDS, temperature, TH, BOD, DO, oil and grease helps to promote good environment for coliform bacteria near discharge sites of industrial waste (Chatterjee et al. 2010). In India, many other studies on river water quality also depict high organic activities due to contamination of industrial, agricultural, or urban sewage to the river water (Chandra et al. 2006; Sundaray et al. 2006; Sharma et al. 2008). In the pre-monsoon season depletion of values of all physicochemical parameters remarkably changed the quality of river water in the study area. Difference of mean values has been statistically analysed by ‘t’ test and it indicates that there are significant differences between values of all parameters in two seasons. Values of PI in post- and pre-monsoon seasons also reject the null hypothesis at a significant level of 0.01% and 0.05%. The main cause of this change is ‘lockdown’ due to the COVID-19 pandemic in the world as well as in India. The lockdown process had started from 25th March 2020 and it continuously acted till 31st July 2020 in India. This lockdown scenario badly affected the world’s economic activity but it helped the environment to get rid off of pollution load temporarily (Masood et al. 2016). Closing of industries near the river banks in this period helped to lower the discharge of industrial waste to the main riverbed. It promotes level of DO in river water by decreasing bacterial activities. But uninterrupted power supply from Durgapur thermal power station (S8) released hot wastewater regularly in this period. Thus, high temperature of river water at this site leads to the highest growth of TVC, TC, and FC than the other sample sites. Mean values of coliform bacteria indicate a lower number of colony formation in the pre-monsoon season than post-monsoon season because of less supply of chemicals and organic wastes to the river water during lockdown period.

6 Environmental Management Planning

The great dilemma of public health and economic progress in the time of the COVID-19 pandemic brings a new normal life to the world. ‘Unlock’ process has started in most of the countries as well as in India to restore the economic sector. ‘Unlock 1’ phase started from first August 2020 in India and many industrial sectors restarted their production all over the country. But the major problem is the absence of proper control of sewage or industrial waste discharge to the environment in most of the developing countries like India (Mukherjee et al. 2020). Industrial effluents and sewage supply various nutrients to microbes and it favourably grows in surface water. This led to faecal contamination and threatened public health (Strauss 1996). Water quality control is the key challenge to the government for the management of poorly treated or untreated wastewater discharged from different anthropogenic sectors on a temporal or spatial scale.

From the present study, it is clear that the environmental quality of river Damodar is highly affected by numerous waste materials supplied by nearby industries to the river bed at different points or locations. It leads to acceleration of bacterial growth and depletes the health of river water. No one parameter is recorded safe for drinking purposes in any sample during post-monsoon season. High pollution load increased faecal contamination such as E. coli in river water. Many studies reported that bacteriological activities of other rivers in India are lower than river Damodar especially of coliform types (Chatterjee et al. 2010). River Ganga in near its origin has a lesser amount of microbial load than river Damodar (Sood et al. 2008). Therefore, it is very necessary to take different measures to control the pollution of river Damodar in its industrial catchment. Here, some strategies are presented to mitigate water pollution and they could be applied for environmental restoration.

1. The National River Action plan should be regulated properly to control and for the treatment of discharged materials by nearby industries or municipalities to the river water.
2. Wastewater contains a high amount of pathogenic contaminants and in most cases, polluted water is drained directly without proper treatment. Thus, a suitable investigation is needed to identify those industries engaged with this practice.
3. Various industries such as chemical, cement, coal washery, tannery, steel plant, thermal power station, and so on, used a high volume of freshwater for their activities and release toxic and organic polluted wastewater to the environment. Strong regulation should be implemented for controlling these activities.
4. Effluent Treatment Plant (ETP) should be installed by the Government or private farms in various industries for the treatment of wastewater before releasing to the environment.
5. Time to time assessment of river water quality will very helpful to sustainable management of water resources.
6. Regular monitoring of microbial activities (coliform count) should be practiced by management bodies.
7. Recreational activities like community bathing; religious practice, and so on, should be checked by Government bodies to control microbial growth in river water.
8. Water (Prevention and control of pollution) Act, 1974 and The Damodar Valley Corporation (Prevention of pollution of water) Regulation act, 1948 should be taken effectively to meet its objectives properly.
9. Various scientific methods and machines such as probiotic strains, bacteriophages, and so on, should be introduced to prevent faecal contamination through drinking water.
10. Furthermore, awareness of people to the environment is a big solution of water resource management. Various awareness programmes for the development of health-related consciousness and water contamination should be implemented by local governing bodies or policymakers.

7 Conclusion

The physicochemical and biological assessment of river Damodar between Panchet dam and Damodar barrage were conducted in two different seasons at 11 sampling points. First investigations in December 2019 as post-monsoon season depicts all parameters were higher than its standard limit of drinking purpose. PI value was recorded highest in S8 (1.93) in this season. S9 (1.90) indicates also a higher value of PI at the same time. Mixing of hot wastewater, high TDS, EC, pH, Cd, Cr, BOD, oil and grease led to high pollution load at these sites and thus microbial activities (TVC, TC, and FC) also found their highest colonies at these discharge sites. S1 indicates the lowest PI value (1.16) due to a very low mixing of chemical effluents from the coal mining fields. Therefore, the microbial load was also found very low in this sample water. Second investigation was conducted in May 2020 as pre-monsoon season and it reveals drastic changes in concentration of all parameters in the sample water because of the 'lockdown' phase in the COVID-19 pandemic situation. Closing of all industries and very low mixing of wastewater noticeably improves the water quality of river Damodar. Low mixing of organic matters helps to decrease microbial load and increase DO in the river water. In this season highest PI was found from the S8 sample (1.13). Discharge of hot wastewater from the thermal power plant and the high downstream load of other parameters helps to sustain microbes in this site than other sites. After the 'unlock' process of the socio-economic sector, it will be a great challenge to sustain the water quality of river Damodar in this study area without proper mitigation or management of industrial wastewater before discharge to the environment. Thus, suitable planning and awareness of common people should be practiced through the results of the present study by the policymakers of the society.

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Chapter 13
Assessment of Water Pollution and Aquatic Toxicity of the Churni River, India

Biplab Sarkar, Aznarul Islam, Pravat Kumar Shit, and Susmita Ghosh

Abstract The present study deals with the spatiotemporal variation of water pollution of the Churni River based on selected physicochemical parameters and metals present in water samples. The water quality and aquatic toxicity of all the water samples have been measured using the weighted arithmetic water quality index and aquatic toxicity index. The results reveal that the upper stretch of the river is more polluted compared to the lower stretch. Moreover, the quality of water at both stretches has become highly deteriorated in the pre-monsoon season compared to the monsoon and post-monsoon seasons. The aquatic toxicity index of most of the samples depicts that water is suitable only for tolerant fish species. Moreover, a multivariate statistical technique has been applied to the physicochemical parameters of water which suggests that anthropogenic influx such as industrial wastewater, inorganic nutrients from agriculture are playing a significant role in inducing the pollution of the Churni River. Moreover, the study has also found that natural forcing (e.g. eastward tilting of Bengal delta) and anthropogenic factors (e.g. the construction of Farakka barrage and road-stream crossings) are also influencing the water quality of the Churni River.

Keywords Aquatic toxicity index · Water pollution · Critical environmental flow · Point and non-point sources · Churni River

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1 Introduction

The river is playing a significant role in providing diversified habitats for countless freshwater species (Edwards and Twomey 1982). In the era of 'Anthropocene', the pristine river having a length of more than 100 km is rare to find. The industrial revolution, urbanisation, modern agriculture and rapid land use and land cover change, etc. have been instrumental to induce river water pollution (Dwivedi et al. 2018; Ding et al. 2016; Kibena et al. 2014). It is documented that the rivers passing through the deltaic region are more vulnerable to pollution. Moreover, from the perspective of hydrological flow, rivers which are facing a declining trend in discharge are more prone to water pollution and suffering from severe ecological problems (Shivoga 2001; Sarkar and Islam 2020). Therefore, measuring the pollution level for developing planning for river restoration and management is a vital challenge to the researchers. Pollution of river water is investigated from multiple perspectives: general use (Bora and Goswami 2017; Alves et al. 2018; Ahmed et al. 2017; Medeiros et al. 2017), drinking water (Wu et al. 2018; Parinet et al. 2004; Boyacioglu and Boyacioglu 2008; Zhang et al. 2009), irrigation water (Sarkar and Islam 2019) and riverine ecology (Van Vuren et al. 1994; Wichert and Rapport 1998; Aminiyan et al. 2018; dos Santos Simões et al. 2008; Koukal et al. 2004). Moreover, many researchers have studied the potential sources of water pollution. For example, Wang et al. (2011) studied the effect of anthropogenic activities on chemical contamination of Grand canal, China. Similarly, Stanley and Preetha (2016) studied the effect of agriculture on nutrients pollution in the river. Furthermore, many studied have focused on the role of road-stream crossing and basin land use and land cover change on the pollution of the river (Sarkar et al. 2020). Therefore, diversified knowledge is continuously sprouting on river water pollution. Moreover, the outcomes of the pollution research have immensely benefitted the planners and policy-makers to successfully restore many rivers and its ecosystem across the world.

The Churni, an important river of the Bengal delta, has undergone multiple pressure: declining trend in discharge, mixing of industrial wastewater, pesticides from on-bed agriculture and urban sewage, cutting of river beds and banks for brick kiln industry, road-stream crossing, etc. Among these, industrial water pollution has become a major problem for the ecological degradation of the river. The Carew and Company located at Darshana in Bangladesh while manufacturing sugar, wine and chemicals release a huge amount of wastewater into the river (Sarkar and Islam 2020). Consequently, the entire course of the river gets polluted and aquatic species especially fish respond critically to this toxic environment. The livelihood of the fishermen residing along the bank of the Churni River has become vulnerable due to the drastic fall of fish production. Therefore, this polluted river has come under scientific evaluation. Few scholastic works are found to address the quality of the Churni River water. For example, Chakrabarty and Das (2006) investigated the ecological effect due to the pollution of the Churni River. Moreover, Panigrahi and Bakshi (2014) and Bakshi et al. (2016) had also studied seasonal variability in the availability of fish species in the Churni River. Besides, Sanyal et al. (2015) studied the role of industrial wastewater discharge on the chromium concentration at the lower stretch of the Churni River. Besides, Sarkar and Islam (2019) analysed the suitability of the Churni River for irrigation use. However, the available literature suggests that the perspectives of spatio-temporality of water pollution and its relation with the aquatic toxicity for the river Churni are not explored. Therefore, the present study intends to address the spatial and seasonal variation of water quality, the level of toxicity and to critically analyse the natural and anthropogenic factors responsible for the pollution of the river.

2 Study Area

The present study has been executed on the Churni River, which is the continuation of the lower part of the Mathabhanha River. The Mathabhanha River takes off from the Padma River (24° 03′ 19″ N and 88° 24′ 21″ E) in Bangladesh. Therefore, the Padma River has a direct influence on the hydrogeomorphological entity of the Churni River. The river runs for ~56 km through four community development blocks (C.D. Blocks): Krishnaganj, Hanskhali, Ranaghat-I and Ranaghat-II (Fig. 13.1). The pollution character of the Churni River and its eventual effect on the aquatic ecosystem has been controlled by many natural and anthropogenic factors. The Mathabhanga-Churni is a transboundary river that crosses and re-crossing the international boundary between India and Bangladesh several times. There is a long international dispute between India and Bangladesh regarding the pollution of the Churni River. The frequent release of a huge volume of industrial wastewater in the Bangladesh part of the Mathabhanga River has polluted the entire downstream part of the Mathabhanga-Churni River. Therefore, the livelihood of many people dependent on the fishing activities of the Churni River has become paralysed. Since the last decade, people have started raising their voices against industrial pollution to make the river pollution free.

3 Database and Methodology

3.1 Database

For assessing the water quality and aquatic toxicity, 168 water samples have been considered (84 each from the Majhdia located at the upper stretch of the river and Ranaghat located at the lower stretch of the river). Again, for assessing the seasonality in water quality all the water samples have been classified into three categories representing pre-monsoon (February to May), monsoon (June to September) and post-monsoon (October to January) season. The parameters considered for water quality analysis are pH, electrical conductivity, total dissolved solids, total

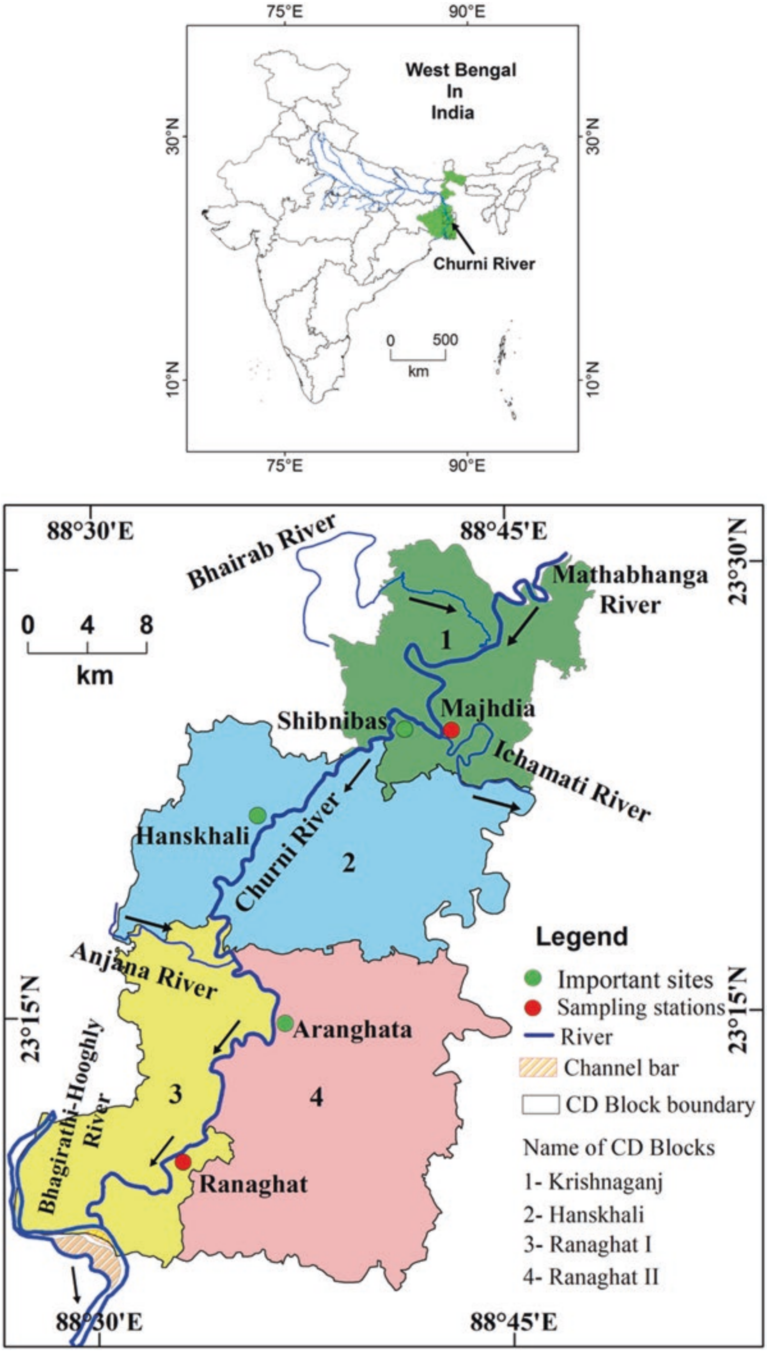

Fig. 13.1 Location of the study area

suspended solids, total hardness, chloride, dissolved oxygen, BOD, sulphate and total alkalinity. Besides, the assessment of aquatic toxicity requires few more parameters which are ammonium, phosphate, potassium turbidity, potassium, fluoride, zinc, chromium, copper, lead and nickel. Furthermore, searching for the role of critical environmental flow and changes in channel morphology on water quality, discharge, velocity, channel width and depth have been measured considering 47 cross-sections. Moreover, for assessing the role of road-stream crossing on the channel hydraulics, pier geometry (diameter, nose shape, pier depth below water) and geometry of dais have been measured. Besides, the area of the on-bed agriculture has been measured by direct field survey and area measurement tool by Google Earth Pro software.

3.2 Methods

3.2.1 Measuring Water Quality

For assessing the general water quality of the Churni River, a water quality index has been applied which is calculated based on the weighted arithmetic method and therefore called as weighted arithmetic water quality index (WAWQI) (Brown et al. 1970). The index is widely accepted because it includes those variables which are significant, reliable and available. This index considers ten parameters which are pH, electrical conductivity, TDS, TSS, total hardness, chloride, BOD, DO, sulphate and total alkalinity. The algorithm for computing the index is mentioned in Eq. (13.1).

$$WAWQI = \frac{\sum Q_n W_n}{\sum W_n} \tag{13.1}$$

where Q_n indicates the water quality rating of the nth parameter while Wn is the unit weight of the nth water quality parameter. The Q_n is computed using Eq. (13.2).

$$Q_n = \frac{(V_n - V_i)}{(V_s - V_i)} \times 100 \tag{13.2}$$

where V_n is the actual concentration of the nth parameter and V_i is the ideal value of the parameter. In this regard it should be mentioned that V_i is 0 for all parameters, except for pH ($V_i = 7$) and DO ($V_i = 14.6$ mg/L). V_s is the standard permissible value for the nth water quality parameter. Unit weight (W_n) which is essential for this index has been calculated using Eq. (13.3).

$$W_n = \frac{k}{V_s} \tag{13.3}$$

where k is the constant of proportionality and it is calculated using Eq. (13.4).

$$k = \frac{1}{\sum \frac{1}{V_s} = 1,\ 2 \ldots\ldots\ldots, n} \tag{13.4}$$

The unit weights assigned to all the respective parameters have been shown in Table 13.1. Finally, the values of WAWQI are divided into five categories: (a) 0–25 as excellent, (b) 26–60 as good and both can be used for drinking, irrigation and industrial; (c) 51–75 as poor and possibly used for irrigation and industrial; (d) 76–100 as very poor and possibly used only for irrigation; (e) above 100 as unsuitable for drinking and fish culture.

3.2.2 Measurement of Aquatic Toxicity

The index of aquatic toxicity was developed by Wepener et al. (1992) for evaluating the toxic effects of selected water quality parameters on aquatic lives, particularly on the growth and physiological activities of different fish species. Of late, this technique is widely applied by researchers for assessing the ecosystem vulnerability of polluted rivers in different parts of the world (Wepener et al. 1999; Gerber et al. 2015). This technique requires two procedures (a) development of sub-index curve and (b) derivation of aggregation function for final index score (Fig. 13.2). The sub-index values have been calculated based on the respective equations developed by Wepener et al. 1992 (Table 13.2). Finally, to get the ATI scores, Solway modified the unweighted additive aggregation function (House and Ellis 1981) has been computed using Eq. (13.5).

Table 13.1 Relative weights (Wn) of the parameters used for WQI determination (all the parameters are in milligrams per litre except pH and EC (μS/cm)

Parameter	Bureau of Indian Standards or BIS standard (V_s)	Unit weight (W_n)
pH	6.5–8.5	0.215
Electrical conductivity	300	0.0061
TDS	500	0.00366
TSS	500	0.00366
Total hardness	300	0.0061
Chloride	250	0.00732
DO	5	0.366
BOD	5	0.366
Sulphate	150	0.0122
Total alkalinity	120	0.01525
		$\sum$Wn = 1.001

Source: Bora and Goswami (2017)

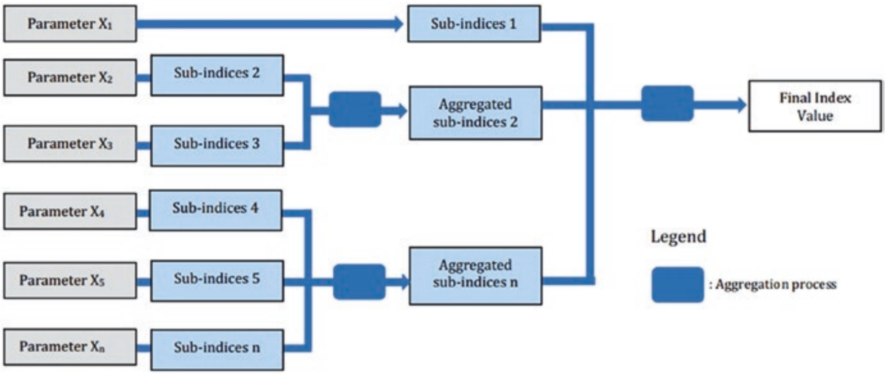

Fig. 13.2 Systematic process for developing water quality index

Table 13.2 Equations for the determination of index rating values for ATI

Parameters	Index rating equation
Dissolved oxygen	0 ≤ DO ≤5; y = 10 (DO) 5 ≤ DO ≤6; y = 20 (DO) − 50 6 ≤ DO ≤9; y = 10 (DO) + 10 DO >9; y = 100
pH	y = 98 exp. [−(pH − 8.16)^2(0.4)] + 17 exp. [−(pH − 5.2)^2 (0.5)] + 15 exp. [− (pH − 11)^2 (0.72)] + 2
Nickel	y = −c in(a (Ni + b)) + d a = 1; b = −10; c = 28; d = 211
Fluoride	y = −c in(a (F + b)) + d a = 0.001; b = 2.5; c = 71; d = −235
Lead	y = −c in(a (Pb + b)) + d a = 0.1; b = −30; c = 27; d = 148
Cupper	y = −c in(a (cu + b)) + d a = 1; b = −18; c = 26; d = 180
Chromium	y = −c in(a (Cr + b)) + d a = 0.1; b = 150; c = 40; d = 210
Zinc	y = −c in(a (Zn + b)) + d a = 0.001; b = −20; c = 22; d = 16
Potassium	y = a exp. $^{-b\,(k)}$ + c a = 150; b = − 0.02; c = −8
Turbidity	y = −c in(a in(NTU) + b) + d a = 0.001; b = 30; c = 220; d = −689
Ammonium	0.02 ≤ NH_4^+, y = 100; 0.02 < NH_4^+ ≤ 0.062 y = −500(NH_4^+) + 110 0.062 < NH_4^+ ≤ 0.5 y = 40/(NH_4^++0.65)^2
Phosphate	y = a exp.(P)b a = 100; b = −2.4

Source: Wepener et al. (1992)

$$ATI = \frac{1}{100}\left(\frac{1}{N}\sum_{i=1}^{n} qi\right)^2 \tag{13.5}$$

where ATI = final index score; *qi* = the quality of ith parameter (the value ranging from 0 to 100); and *N* is the number of determinants involved in this indexing system.

Moreover, apart from the modified unweighted additive aggregation function, the minimum operator function has been computed using Eq. (13.6).

$$I = \text{minimum}\left(Isub_1 Isub_{2,\ldots\ldots\ldots\ldots\ldots,} Isub_n\right) \tag{13.6}$$

Finally, the ATI value interprets the suitability of water quality for all fish species (Table 13.3).

3.2.3 Multivariate Statistical Technique

In the realm of multivariate statistical analysis, PCA is a sophisticated technique widely used to reduce the dimensionality of large data set with minimum loss of original information. Many researchers have used this technique for assessing both the surface and groundwater quality. Moreover, this technique is really helpful for identifying potential sources responsible for pollution. Technically, it can be said that through this technique eigenvalues and eigenvectors are extracted from the covariance matrix of original variables. In this process uncorrelated (orthogonal) variables commonly known as PCs are obtained multiplying the original correlated data with the eigenvectors. So, PCs are the result of the linear combination of original variables and eigenvectors, a list of coefficients (loadings or weightings). Moreover, eigenvalues are very important in confirming the significance of PCs. Thus, PCs having the highest eigenvalues are the most significant. The mathematical algorithm of PCA is expressed using Eq. (13.7).

Table 13.3 Description of the range of sub index value (*I sub*) and ATI scale

Range of sub index	Indication	ATI Scale	Interpretation
$100 \geq I\ sub \geq 80$	Eminently suitable for all use	60–100	Indicates water of suitable quality for all fish life
$80 > I\ sub \geq 60$	Suitable for all use	51–59	Indicates quality of water suitable only for hard fish species.
$60 > I\ sub \geq 40$	Some uses may be compromised	0–50	Indicates quality of water which is totally unsuitable for normal fish life
$40 > I\ sub \geq 20$	Unsuitable for several uses		
$20 > I\ sub \geq 0$	Totally unsuitable for many uses		

Source: Smith 1990

$$Zij = \alpha_{i1} X_{1j} + \alpha i_2 X_{2j} + \quad \alpha_{i3} X_{3j} + \alpha_{im} X_{mj} \tag{13.7}$$

where Z = component score; α = component loading; X = measured value of variables; i = component number; j = sample number; m = total number of variables.

4 Results

4.1 Water Characteristics of the Churni River

The statistical summary of the values recorded for pH, EC, TDS, TSS, TH, chloride, DO, BOD, TA and sulphate at both the upper and lower stretches of the river and during the pre-monsoon, monsoon and post-monsoon seasons are presented in Table 13.4. The pH of all water samples represents the normal range for surface water system as well as BIS desirable limit. The average value of EC at the upper stretch is found as 624 µs/cm for pre-monsoon, 381.14 µs/cm for monsoon, 551.75 µs/cm for post-monsoon and at the lower stretch 641.32 µs/cm for pre-monsoon, 357.29 µs/cm for monsoon, 554.18 µs/cm for post-monsoon. Therefore, all the EC values have crossed the BIS desirable limit (300 µs/cm) and the highest is recorded for pre-monsoon for both the upper and lower stretches. Besides, similar phenomena have also been observed for DO, BOD and TA. DO is the most threatening parameter and the concentration of it in all the seasons and at both, the stretches are found critically low which cannot support to functioning the normal physiological activities of the fish species, even the DO values < 1 are recorded in few samples at the upper stretch which is an indication of fish-kill. Regarding BOD, all the average values except post-monsoon are recorded significantly above the BIS desirable limit (5 mg/L). Moreover, all the average values of TA are also found to exceed high and not desirable as per the BIS recommendation. Contrastingly, all the values for TDS, TSS, TH, chloride and sulphate are within the BIS desirable limit. Moreover, few parameters are observed to have a statistically significant difference between the upper and lower stretches.

4.2 Water Quality and its Spatiotemporality

The results of the WAWQI (annual average) reveal that the quality of the Churni River ranges from very poor to unsuitable at both the upper and lower part from the year 2011 to 2017 (Fig. 13.3a). It has also been observed that the value of WAWQI in the upper stretch is comparatively higher than that of the lower stretch every year except 2014. In the case of Majhdia, the most significant factors responsible for water quality deteriorating have been DO and BOD because they have the highest contribution to $QnWn$ compared to other parameters. In more detail, in the years of

Table 13.4 Spatiotemporal variation of concentration of water quality parameters

Parameters	Majhdia			Ranaghat		
	Pre-monsoon (mean ± SD)	Monsoon (mean ± SD)	Post-monsoon (mean ± SD)	Pre-monsoon (mean ± SD)	Monsoon (mean ± SD)	Post-monsoon (mean ± SD)
pH	7.59 ± 0.40	7.68 ± 0.39	7.61 ± 0.41	7.67 ± 0.28	7.58 ± 0.29	7.65 ± 0.27
EC (μS/cm)	624.14 ± 67.87	381.14 ± 173.14	551.75 ± 130.67	641.32 ± 56.31	357.29 ± 151.38	554.18 ± 140.07
TDS (mg/L)	402.07 ± 54.60	259.86 ± 136.42	351.64 ± 80.75	393.64 ± 65.06	250.86 ± 110.69	354.79 ± 92.12
TSS (mg/L)	49.79 ± 71.92	71.93 ± 79.83	43.64 ± 49.83	31.75 ± 27.52	47.02 ± 51.31	49.55 ± 58.55
TH (mg/L)	300.50 ± 48.46	185.61 ± 81.73	247.14 ± 77.98	305.86 ± 57.60	184.32 ± 75.89	245.11 ± 83.26
Chloride (mg/L)	12.90 ± 15.90	6.02 ± 3.32	8.28 ± 1.96	14.80 ± 2.60	8.65 ± 3.66	10.58 ± 2.57
DO (mg/L)	1.86 ± 2.42	3.48 ± 2.49	2.76 ± 3.74	3.78 ± 1.67	3.41 ± 1.85	4.04 ± 3.11
BOD (mg/L)	11.10 ± 13.61	6.49 ± 6.69	4.73 ± 3.12	5.57 ± 3.03	5.53 ± 3.76	4.61 ± 3.05
TA (mg/L)	322.29 ± 63.18	204.93 ± 98.05	290.21 ± 90.26	324.57 ± 81.04	178.14 ± 80.08	285.50 ± 96.41
Sulphate (mg/L)	4.37 ± 2.49	6.63 ± 4.45	4.68 ± 2.32	5.18 ± 4.09	7.39 ± 6.09	5.69 ± 3.43

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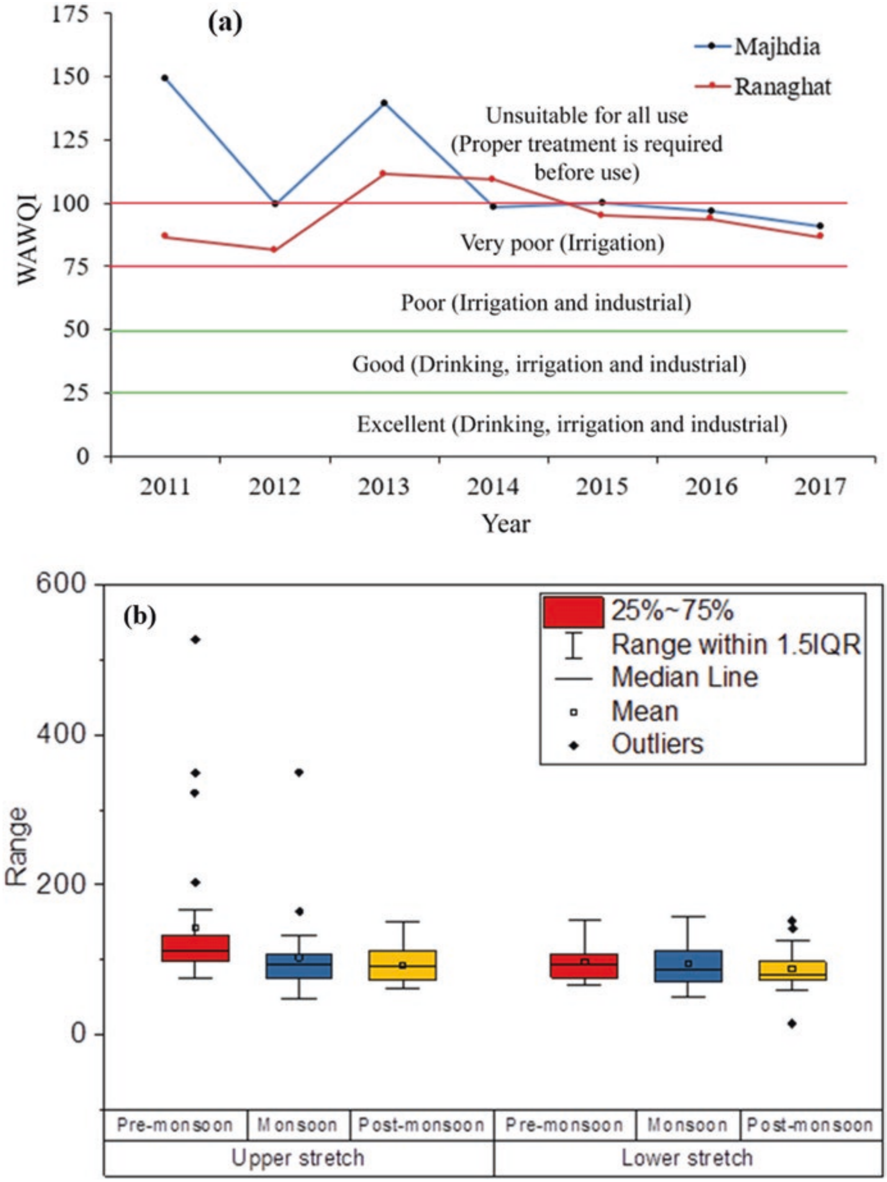

Fig. 13.3 Spatiotemporality in water quality (**a**) year-wise distribution of WAWQI and (**b**) seasonal pattern of WAWQI

2012, 2014, 2016 and 2017 the DO has become the leading factor while the other years (2011, 2013 and 2015) BOD has become the leading factor having the highest value of QnWn. Next to DO and BOD, pH and total alkalinity have played an important role in lowering the upstream water quality. A similar observation has also been found for the lower stretch. At Ranaghat, in the years of 2011, 2012, 2016 and 2017 the DO has become the leading factor while the other years (2013, 2014 and 2015) BOD has become the leading factor having the highest value of $QnWn$.

Regarding the seasonal pattern of WAWQI, it has been observed that at the upper stretch 67.86% of samples come under the unsuitable category followed by very poor (28.57%) and poor (3.57%) in the pre-monsoon season while in monsoon it is highest for very poor followed by unsuitable and poor category and in case of post-monsoon greater samples come under unsuitable followed by very poor and poor category (Fig. 13.3b). A similar trend has also been found for the lower stretch of the river though relatively better water quality has been observed at the lower stretch of the river (Table 13.5). Therefore, the index reveals that both the stretches of the river pre-monsoon season exhibit the worst water quality compared to the monsoon and post-monsoon seasons. Moreover, it has also been observed that no samples come under the excellent and good quality category except for post-monsoon at a lower stretch (3.57%) (Table 13.5). Regarding the spatiality in seasonality, the results of ANOVA at 0.05 significance level and 54 degrees of freedom exhibit that there is a statistically significant difference of water quality between the upper and lower stretch only in the pre-monsoon season ($p = 0.019$) and other seasons nullify the difference ($p = 0.47$ for monsoon and $p = 0.44$ for post-monsoon).

Table 13.5 Spatiotemporal variation of water quality

Range of WQI	Water quality status	Possible uses	Majhdia			Ranaghat		
			Pre-monsoon (%) ($n = 28$)	Monsoon (%) ($n = 28$)	Post-monsoon (%) ($n = 28$)	Pre-monsoon (%) ($n = 28$)	Monsoon (%) ($n = 28$)	Post-monsoon (%) ($n = 28$)
0–25	Excellent	Drinking, irrigation and industrial	0	0	0	0	0	3.57
26–50	Good	Drinking, irrigation and industrial	0	0	0	0	0	0
51–75	Poor	Irrigation and industrial	3.57	21.43	28.57	21.43	32.14	28.57
76–100	Very poor	Irrigation	28.57	42.86	39.29	35.71	39.29	46.43
Above 100	Unsuitable for drinking and fish culture	Proper treatment required before use	67.86	35.71	32.14	42.86	28.57	25.00

Computed by authors, 2021

4.3 PCA of the Physicochemical Parameters

PCA has been applied separately to the data sets representing pre-monsoon, monsoon and post-monsoon seasons of both the stretches of the river. Regarding the upper stretch, 63.04% of the data variation is explained by two principal components (PC) for pre-monsoon having high positive loadings on TDS, TSS and chloride in PC1 and EC in PC2 while significant negative loading on DO in PC1 and Sulphate in PC2 (Fig. 13.4a). For monsoon, 64.31% variance is explained by PC1 and PC2 where significant positive loadings are found to have on EC, TDS, TH, TA in PC1 and DO in PC2 (Fig. 13.4b). Moreover, for post-monsoon PCA 58.92%

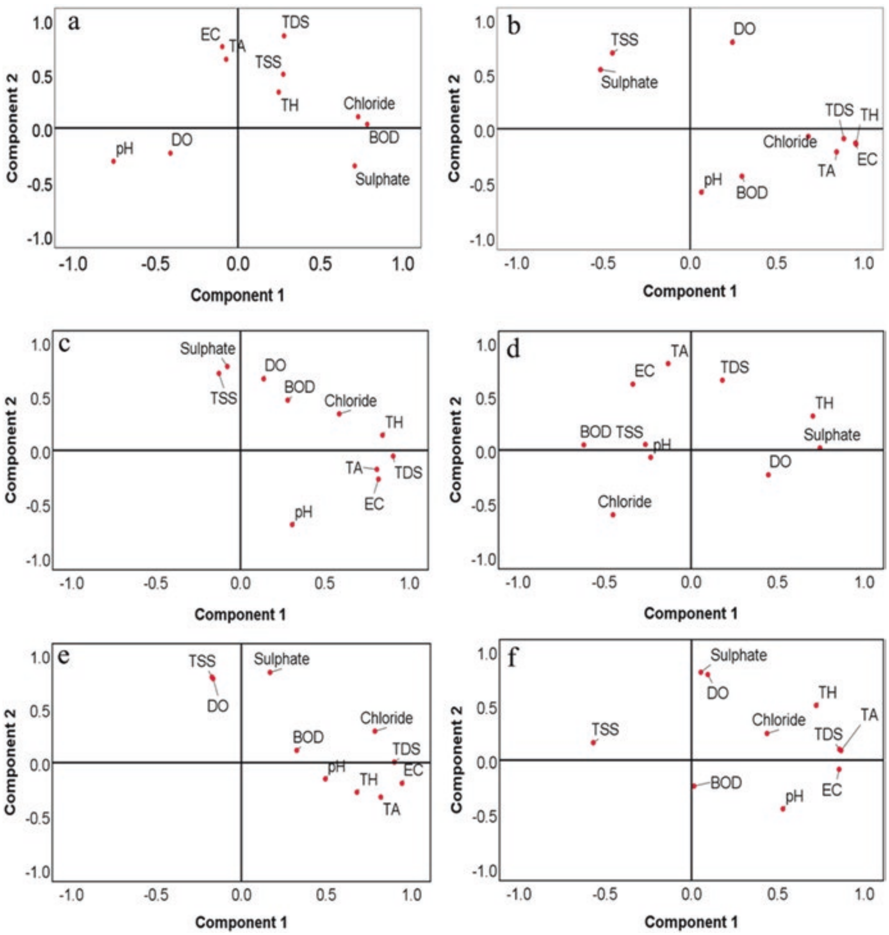

Fig. 13.4 Principal component analysis (**a**, **b**, **c**) for pre-monsoon, monsoon and post-monsoon at the upper stretch; (**d**, **e**, **f**) for pre-monsoon, monsoon and post-monsoon at the lower stretch of the river

variance is explained by PC1 (having high positive loadings on EC, TDS, TA) and PC2 (high positive loadings on sulphate) (Fig. 13.4c). Therefore, all the seasons are representing similar parameters with significant loadings except negative loading on DO in PC1 for pre-monsoon. Therefore, the high positive loadings of TDS, EC, TSS and chloride are representing wastewater pollution from domestic, industrial sources while negative loading on DO indicates oxygen-consuming organic pollution from municipal sewage and industrial wastewater and nutrient pollution from agriculture sources. Similarly, regarding the lower stretch, PC1 and PC2 are explaining 56.17%, 72.65% and 54.41% of the total variance for per-monsoon, monsoon and post-monsoon, respectively. Moreover, their patterns of loading also representing similar kind of pollution sources as retained for the upper stretch of the river (Fig. 13.4 d, e, f).

4.4 Nature of Aquatic Toxicity in the Churni River

The highest ATI value at the upper stretch of the Churni River is recorded as 77.67 in 2013 representing that the water is suitable for all fish life. But, most of the years come under the category of ATI:51–60 indicating that water is suitable only for hard fish species (Table 13.6). Moreover, the value in 2011 represents unsuitable water for normal fish life. The corresponding minimum operator scores also reveal that the quality of water at the upper stretch is strongly influenced by the faecal coliform DO and turbidity. The highest values recorded for faecal coliform and turbidity are 1,100,000 most possible number (MPN)/100 millilitre (mL) and 57 (Nephelometric Turbidity Unit) (NTU), respectively. On the other hand, the minimum DO concentration is recorded as 0.4 mg/L. Besides, the other parameters such as ammonia, phosphate and BOD have their considerable effect on increasing the toxicity level. Moreover, though heavy metals such as copper, chromium and nickel are negligibly present in suspended sediment, during the industrial wastewater discharge these metals play a vital role in inducing the toxic level of water. Therefore, the highest

Table 13.6 Year-wise distribution of ATI values and corresponding minimum operator

	Majhdia		Ranaghat	
Years	ATI score	Minimum operator	ATI score	Minimum operator
2011	48.82	Faecal coliform (5)	60.84	Faecal coliform (8)
2012	55.65	Faecal coliform (7); DO (7)	68.12	Faecal coliform (25)
2013	77.67	Turbidity (10)	63.04	Faecal coliform (6)
2014	55.75	Faecal coliform (4); DO (4)	58.37	Faecal coliform (8)
2015	63.47	Faecal coliform (5)	63.79	Faecal coliform (8)
2016	56.65	Faecal coliform (6)	52.42	Faecal coliform (6)
2017	59.90	Faecal coliform (7)	63.04	Faecal coliform (7)

Computed by authors, 2021.

concentration of copper, chromium and nickel are observed as 372.5 μg/L, 213.8 μg/L and 40.4 μg/L, respectively, at the upper stretch of the river.

A little contrasting nature of ATI has been observed for the lower stretch of the river where the highest ATI score has been observed as 68.12 with the corresponding minimum operator score of 25 for faecal coliform. Moreover, most of the years come under the category of ATI:60 –100 indicating the water is suitable for all fish. Again, faecal coliform in all the years has been retained as a single parameter highly responsible for increasing the toxicity level and the highest values recorded for faecal coliform and turbidity are 900,000 MPN/100 mL. The other parameters, similar to the upper stretch, have also a considerable effect on water quality at the lower stretch but the intensity is not as much as present in the upper stretch.

4.5 Risk Potential of Freshwater Fish Species

The average concentration of ammonia nitrate and BOD in all seasons and at both the stretches of the river are found above the desirable limit (Table 13.7). Besides, the DO follows the trend opposite to the ammonia nitrate and BOD. At all the seasons and both the reaches the average concentrations of DO are critically low and it poses threat to the fish species to survive. Therefore, their elevated concentration gives birth to many diseases in the fish species. The experiment of fish health conducted by Das et al. (2014) confirmed that due to acute water pollution different fish species at the upper stretch of the Churni River are suffering from many chronic diseases like hypochloraemia, hyperglycaemia, triglycerides (as stress on blood chemistry), enhanced cortisol, hyperplasia, hypertrophy and aneurism (as cellular stress) etc. while only hyperglycaemia and cortisol were strongly detected among the fishes of lower stretch. Moreover, from the study of Das and Chakrabarty (2007) and Sarkar and Islam (2020), it has been observed that the maximum weight of different fish captured has been decreased significantly. Besides, Sarkar and Islam (2020) also found the effect of water pollution on freshwater fish population and productivity. For example, the productivity of few fish species like *Labeorohita*, *Puntiussaranasarana*, *Puntiusticto*, *Puntiussophore*, *Mystusvittatus* and *Channastriatus* are decreasing above 10% per year while a fall rate of ~ 5% per year was recorded for other species like *Heteropneustesfossilis*, *Clariasbatrachus*, *Channa punctate, Glossogobiusguiris, Channamarulius, Amphipnouscuchia, Channaorientalis*. Therefore, the existing literature and the present study reveal that the aquatic environment is not suitable for general physiological activities and fish growth.

Table 13.7 Seasonal variation of water quality parameters and their possible effect on fish species

Parameters	Desirable limit	Majhdia			Ranaghat			Possible effects on fish species
		Pre-Monsoon (Mean ± SD)	Monsoon (Mean ± SD)	Post-Monsoon (Mean ± SD)	Pre-Monsoon (Mean ± SD)	Monsoon (Mean ± SD)	Post-Monsoon (Mean ± SD)	
Ammonia (mg/L)	< 0.1 mg/L	0.34 ± 0.29	0.21 ± 0.12	0.27 ± 0.17	0.41 ± 0.30	0.24 ± 0.28	0.32 ± 0.24	Fish of the river may suffer from gill damage, osmoregulatory imbalance, slow destruction of mucous producing membranes etc. by ammonia poisoning
Nitrate (mg/L)	0.2 mg/L	0.47 ± 0.41	0.61 ± 0.49	0.72 ± 0.42	0.62 ± 0.45	0.69 ± 0.41	0.82 ± 0.37	High concentration of nitrate in water converts hemoglobin to methemoglobin by oxidization process and makes the blood and gill brown and even damage of respiration and nerve system of fish
DO (mg/L)	DO < 5 mg/L	2.22 ± 2.80	3.49 ± 1.79	1.61 ± 1.35	3.80 ± 1.73	3.41 ± 1.86	3.53 ± 0.90	Slow fish growth
BOD (mg/L)	5 mg/L	11.10 ± 13.17	4.71 ± 2.67	4.58 ± 2.42	5.34 ± 3.01	5.53 ± 3.77	4.66 ± 3.17	It kills fish owing to suffocation.

		Majhdia			Ranaghat			
Parameters	Desirable limit	Pre-Monsoon (Mean ± SD)	Monsoon (Mean ± SD)	Post-Monsoon (Mean ± SD)	Pre-Monsoon (Mean ± SD)	Monsoon (Mean ± SD)	Post-Monsoon (Mean ± SD)	Possible effects on fish species
Hardness (mg/L)	75–150 mg/L	297.86 ± 44.74	157.75 ± 62.32	267.80 ± 69.24	303.71 ± 57.81	184.32 ± 75.89	218.14 ± 75.28	The effect of hardness on eggs hatchability and larval viability found that water having hardness 60 mg/L support the highest larval survival (~ 71 %)

Computed by authors, 2021.

5 Discussion

The polluted nature of the Churni River is a response to the critical environmental flow and pollution from a different point and non-point sources. Therefore, the critical examination of environmental flow and related morphological changes and pollution from different points and non-point sources is needed.

5.1 Critical Environmental Flow

The Mathabhanga-Churni River is facing a chronic problem of discharge since the last century. The volume of discharge which was high at the beginning of the twentieth century has been significantly lowered down at the present. For example, during 1915 the average discharge was 3600 cusecs while in 2015 it was only about 1930 cusecs at Hanskhali gauge station (Biswas 2001). In 2018, the discharge investigation reveals that the average discharge throughout the river was ~2000 cusec. Besides, in the discharge hydrograph of 1915 at Hanskhali gauge station, the peak was widened which indicates that the river would get a maximum discharge in all the monsoon months whereas the discharge hydrograph of 2018 at the same station reveals that the peak is very narrow which indicates only in 1 month in the monsoon season the river attains the highest discharge (Fig. 13.3b and d). Similarly, the velocity and specific stream power in the Churni River are recorded exceptionally low at both the upper and lower stretches of the river. For example, the average velocity at the upper stretch has been recorded as 0.25 m/s where the minimum velocity (0.15 m/s) is observed immediately upstream of the Shibnibas road-stream crossing. A similar observation is retained for the lower stretch (average 0.29 m/s; minimum velocity (0.12 m/s) found immediately upstream of the Kalinariyanpur Road and Rail bridge (Fig. 13.5a). Therefore, the low values of Specific stream power have also been recorded for both the upper and lower stretches of the river (Fig. 13.3c). The changes in channel hydraulics have also induced many morphological changes in the river. For example, the average channel width in 1849 was 162.5 m and reduced to 63.5 m in 2007. The average channel depth was 5.7 m in 1900 and reduced to 4.4 m in 2018. Therefore, the significant hydro morphological changes induce the pollution of the Churni River.

The critical environmental flow in the Churni River is controlled by both natural and anthropogenic factors. Regarding the natural processes, eastward tilting of the Bengal delta during the Holocene period and consequent deep incision by the Ganga–Padma River has become responsible for the seasonal beheading of the Mathabhanga-Churni river at its off-take (Rudra 2014). Moreover, the process of disconnection has been accelerated by the rapid sedimentation at its entry point and only in monsoon months, the river gets connected with the Ganga river. Regarding anthropogenic control, the Farakka barrage has become the main driver of lowering the discharge of the Mathabhanga-Churni river. The Farakka barrage was

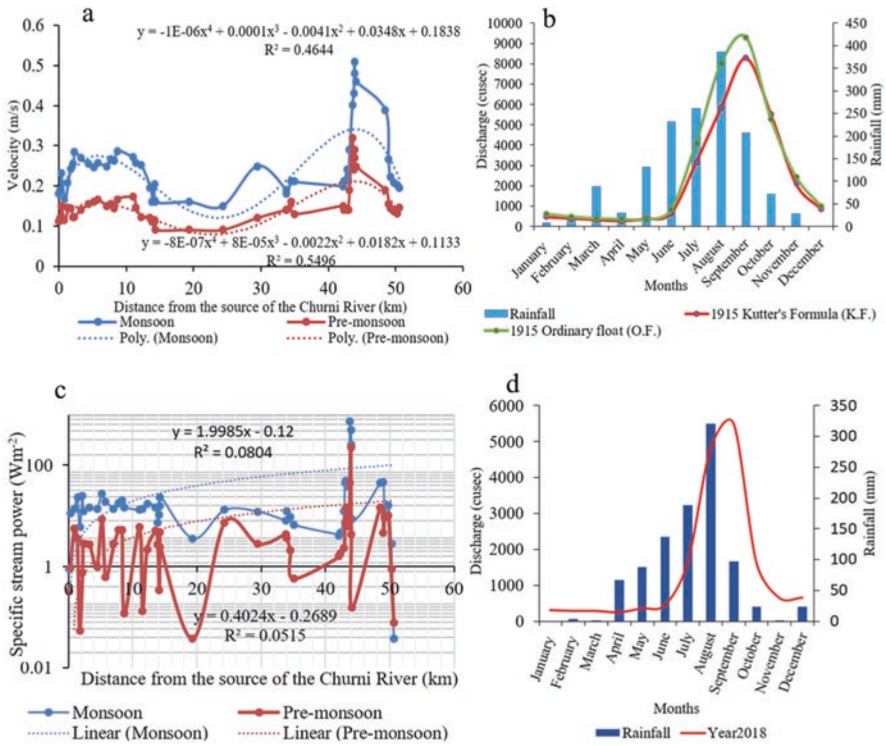

Fig. 13.5 Critical flow in the Churni river (**a**) velocity (**b**) discharge 1915 at Hanskhali (**c**) specific stream power (**d**) pattern of discharge at Hanskhali in the Churni river, 2018

inaugurated in 1975 with an aim to keep the Bhagirathi-Hooghly River navigable throughout the year for the activities of Kolkata port (Islam and Guchhait 2017). In 1996, an agreement was signed between India and Bangladesh for 30 years. According to the agreement India has a right to withdraw the Ganga water flow up to 40,000 cusecs at the Farakka in the dry months (January to May). But, if the availability of water flow at Farakka reduces below 70,000 cusecs, sharing of flow would be divided equally between the countries. Therefore, the Farakka barrage has significantly altered the hydrological pattern in the downstream part of the Ganga River i.e. in the Padma River (Guchhait et al. 2016). Mirza (1997) observed a significant change in the downstream hydrology after the construction of Farakka barrage. The discharge in the monsoon is increased while an alarming decrease has been observed in the dry months' discharge. Rahman and Rahman (2018) also argued that in the pre-Farakka period (1935–1975), the minimum, average and maximum discharge was higher than that of the post-Farakka period (1976–2015) at Hardinge Bridge gauge station (Fig. 13.6). Many other researchers are supporting the opinion that after the construction of the Farakka barrage the dry months' water level every year in the Padma River remains much lower than the bed level of many

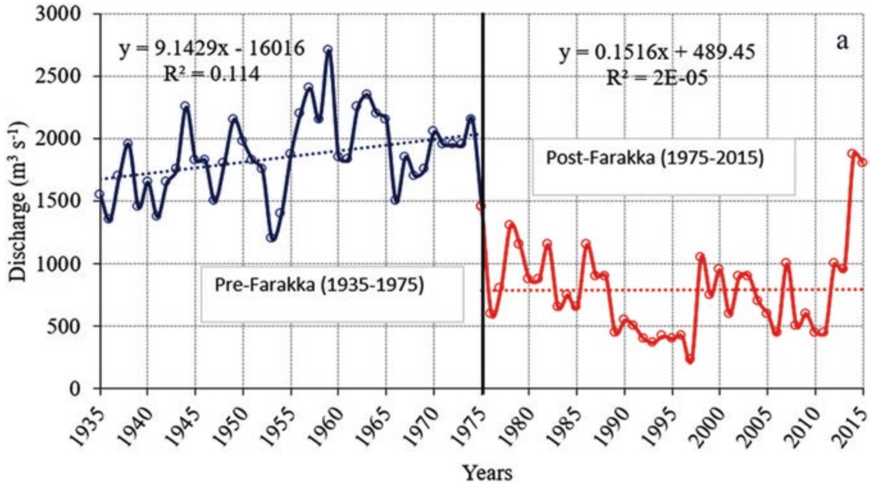

Fig. 13.6 Pattern of discharge in the dry season (January to May) at Hardinge Bridge, the Padma River

distributaries of the Padma River like the Mathabhanga-Churni, Jalangi, Garai, etc. (Sarkar et al. 2020; Gain and Giupponi 2014). Therefore, many distributaries of the Ganga River are facing the chronic problem of discharge unavailability. Consequently, the other problems like a lessening of depth, pollution from point and non-point sources have become prominent in the distributary rivers.

5.2 Point and Non-point Sources of Pollution

Various point and non-point sources of pollution are strongly inducing the pollution of the Churni River (Fig. 13.7a–d). Among the point sources, industrial wastewater discharge into the the Mathabhanga–Churni River by the Carew and Company has made an evolutionary change in the water quality of the Churni river. Being proximity of the industry to the upper stretch, the intensity and exposure frequency of the pollution is high for the upper stretch compare to the lower stretch. During industrial water pollution, the concentration of DO is found many times lower than the average concentration. For example, the average DO concentration is recorded as 3.4 mg/L while during the industrial pollution DO was recorded as 0.4 mg/L which has been lethal for fish life. The measurements of other parameters have also indicated the strong influence of industry on water pollution. Moreover, in a normal situation, the concentration of metals is below the detectable level but during the industrial water pollution, the concentration of the significant metal has been detected at both the stretches of the river. In addition to the industrial role, mixing the untreated urban wastewater through canals is another cause of water pollution. At the lower stretch, many canals are found working for discharging the urban

Fig. 13.7 Multiple perspectives of the Churni River pollution (**a**) black water during the discharge of industrial wastewater from Carew and company, Bangladesh, (**b**) On-bed agricultural practice in the Churni River, (**c**) flow regulation by road-stream crossing, (**d**) unusual growth of water hyacinth

wastewater (Table 13.8) into the river and consequently lowering the quality of the river water. On-bed and off-bed agricultural practices are playing an important role as the non-point sources of pollution. The field investigation records that about 5% area is being used for on-bed agriculture which significantly adds ammonia, nitrate and phosphate into the river as pollutants. The investigation carried out by Sarkar et al. (2020) found that in the agriculture-dominated area the concentrations of ammonia, nitrate and phosphate are high compare to the other part of the river. Moreover, they observed that the effect of on-bed agriculture on water pollution is stronger than the off-bed agricultural practice.

6 Conclusion

The present study has perceived the declining trend of water quality in terms of WAWQI and ATI. The average value of WAWQI in 2011 indicates the water is unsuitable for all use at the upper stretch of the river and very poor for the lower stretch. Moreover, the year 2016 and 2017 also represents very poor water quality at both the stretches of the river. Besides, the ATI score of the majority of the samples

Table 13.8 Major water-polluting industries located within the catchment area of the Churni River

Name and address of the pollution unit	Locational coordinate		Water consumption in kiloliter per day (KLD)		Wastewater generation in KLD		
	North	East	Industrial	Domestic	Industrial	Domestic	Mixed
Amson textile industries Ranaghat municipality	23°10′58.94″	88°33′58.08″	13	0.4	–	–	8
Shree Durga processing and finishing Mills Pvt. ltd.	23°10′48.49″	88°33′50.48″	9.25	1.5	–	–	1.65
Nutan Fulia Tantubay Samabay Samity ltd.	23°10′48.45″	88°33′50.49″	3.6	0.5	3	0.4	0
Tangail Tantujibi Unnayan Samabay Samiti ltd. Santipur	23°10′44.41″	88°33′43.77″	7.2	1.2	6	0.9	0
FuliaTangail Bayan Silpa Samabay Samity ltd. Samabay Sadan	23°10′39.04″	88°33′33.10″	4.2	0.5	3.5	0.42	–
PMA Handweaves private limited	23°10′38.83″	88°33′17.64″	10	1.5	7	1.5	–

Source: River rejuvenation committee, West Bengal (2020) is identified as the quality of water is suitable only for hard fish species. Therefore, the spatiotemporal variation of water quality and aquatic toxicity of the Churni River is supposed to have been highly significant for geomorphologists, ecologists, environmentalists and other academicians for getting adequate knowledge about the pollution of the Churni River. The study has also focused on the natural forcing and anthropogenic processes responsible for pollution. Therefore, this study would be highly beneficial for the regional planners and policymakers for restoring the river. Furthermore, this work would help to develop awareness among the people about pollution. Above all, the analytical contents of this work would provide a soft indication regarding the necessity to take the river restoration works.

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Chapter 14
River Corridor Mapping and Monitoring Using Geospatial Technology

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Abstract River corridor observation can be done effectively at a variety of scales and in challenging environmental environments using remote sensing, which is quickly expanding. The aim of this chapter is to demonstrate the current and future geospatial strategies that can be used to better understand the river corridor. We begin by reviewing existing approaches for river corridor surveillance, which are framed by the context and sizes on which they are implemented. We also do a "horizon check" for potential approaches that could become more prevalent in planning and applications, mentioning examples from both inside and beyond the fluvial domain. Prioritizing process observations and simultaneous multi-sensor data collection is more likely to result in a greater advance in knowledge than just using improved surveying techniques. The advancement of techniques and lower equipment costs have aided testing, administration, and manufacturing uses, encouraging consumers to choose the most appropriate technique from a variety of options.

1 Introduction

River corridors may be broadly specified to include river courses, riparian areas, alluvial plains, and associated fluvial beds, resulting in an overarching classification scheme for management and research (Harvey and Gooseff 2015). River corridors play a vital role in the climate, economy, and culture, but they also pose one of the most adverse environmental threats in the world, creating it important for surveillance to increase our awareness and protect people (Tomsett and Leyland 2019). The view of the river corridor sees hydrological trade flows and the subsequent improvement of biogeochemical processing as crucial to fostering safe stages of

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fluvial metabolism (Helton et al. 2011). Monitoring of watercourses, their ecosystems, banks, and relationship with the local floodplain is needed to monitor the hydromorphological nature of rivers. Conventionally, such knowledge is gathered by field surveys and historical data, and manual interpretation, all of which are prone to human error and arbitrariness. Aquatic ecologists behave identically the phrases micro-, meso-, or macrohabitats, based on the natural processes and related biocoenoses of these multiple geographical units. From an ecological point of view, the various fluvial characteristics or ecosystems, comprising those in the surrounding alluvial plains, experience irreversible hydromorphological and environmental variations due to pressures and fluxes from the catchment or basin scale, including flow and sediments (Hohensinner et al. 2018). Due to various man-induced practices worldwide such as hydropower extraction, rivers are rapidly under threat, and only rare free-flowing rivers exist, most of them in inaccessible areas of the country (Grill et al. 2019).

In comparison, rivers can pose a major threat to those within the vicinity, mainly by floods (Hirabayashi et al. 2013). The most severe natural threat is known as floods, responsible for 43% of all disasters between 1995 and 2015, with flood incidents expected to be more extreme caused by climate change (UNISDR & CRED 2015). Floodwaters pass over levees and into lower areas on the floodplain's top before entering the river at a downstream point. This complex interaction of longitudinal and lateral fluxes through river corridors has far-reaching implications for the ecology and hydraulic conductivity of the downstream aquatic environment (Harvey and Gooseff 2015). However, in terms of water quality, sediment levels, and general biological richness, global waterways are degrading (Vörösmarty et al. 2010). Freshwater fish (Muneepeerakul et al. 2008) and stream ecology in general (Battin et al. 2003) are housed in river networks along with paths to severe waterborne disease (Rinaldo et al. 2017) and zoonotic diseases.

Presently, advancements in geospatial tools have helped to develop strategies for remote sensing-based river corridor research (Piégay et al. 2019; Fryirs et al. 2019). Geospatial tools provide an ample supply of large-scale data that can be used in an analytical and consistent manner to classify the status of rivers. The geospatial view of rivers made it possible to evaluate long-established fluvial geomorphology and river ecology doctrines, like downstream hydraulic geometry (Leopold and Maddock 1953) or the idea of river continuity (Vannote et al. 1980) and, more specifically, patch-oriented architectures (Poole 2002). Furthermore, open-source platforms such as the toolbox for the fluvial corridor were built to facilitate the automatic GIS study of river corridors (Roux et al. 2015). The use of fine spatial resolution data obtained from UAV or advanced satellite sensors has been a strong subject of recent studies in fluvial remote sensing (Demarchi et al. 2016).

We need persistent data on different scales to truly know the river corridor, with remote sensing being the best way to accomplish this, enabling us to validate the hypothesis that has been proposed as well as provide a foundation for our interpretation of the fluvial type. Nowadays, a few research studies investigate the large-scale structure of river channels using open-access datasets. Data-scarce regions, however, have so far been largely overlooked, even though the majority of lingering free curving rivers are situated in those areas. This chapter attempts to illustrate the present and potential approaches used to help us understand the river corridor. We also intend to identify research analysis that incorporates various remote sensing approaches, such that before "horizon scanning" they gain novel insights into river corridors and explore to propose a potential agenda for river corridor remote sensing.

2 Geospatial Application in River Corridor Mapping

Researchers started using early aspects of remote sensing in the twentieth century by analyzing aerial photographs to analyze fluvial morphology and the mechanisms that underpin it (Coleman 1969). Grabowski et al. (2014) provide guidance and advice for evaluating geomorphological transition in rivers over time, which may assist with process-based river conservation and regeneration. The report is organized around a hierarchical spatial structure that situates the scope, as well as its specific geomorphic models and frameworks, in a larger spatial context. Bhunia et al. (2016) want to look at the improvements in the Ganges river courses, as well as the land use/land cover characteristics that go along with it. The geometry of the meander was measured precisely. Over a 21-year period (1989–2010), the history of river course changes as well as improvements in land use/land cover was explored. Bizzi et al. (2016) define cutting-edge remote sensing technology that can be used to measure hydromorphological properties at any of the River Hierarchical Framework's (RHF) spatial scales (i.e., catchment, landscape unit, river section, river reach, sub-reach—geomorphic and hydraulic units). They also present the findings of a survey on the accessibility of satellite data in EU member states, which illustrate the existing ability to extract RHF hydromorphological pointers from fine resolution multispectral satellite data and topographic LiDAR at the regional level throughout Europe. Weissteiner et al. (2016) considered freely accessible data from the European Union to measure pan-European-scale floodplain ecosystems. In a fuzzy logic framework, they integrated topographic analysis and multispectral satellite data and were able to deliver detailed evidence on this fine spatial scale. Mandal et al. (2018) investigated the prominence of Bhagirathi river course variation identification using satellite data and GIS tools in Murshidabad district over a 41-year period (1977–2017). The river course's sequential changes were delineated using Landsat multispectral scanner (MSS), Thematic Mapper (TM), Enhanced Thematic Mapper (ETM^{+}), and Operational Land Imager (OLI) sensor data from 1977, 1990, 2000, 2010, and 2017. Gartner et al., (2018) present a novel approach for determining areas vulnerable to floods and geomorphic transition, as well as locations critical for riparian ecology and water quality, using publicly available geospatial data. The system establishes a complete river corridor by integrating five process units: (1) flood unit, (2) landslide and steep landscape, (3) wetland, (4) channel movement, and (5) riparian ecological unit. Scorpio et al. (2018) provide a geomorphological map of a 115-km section of the Adige valley bottom in northeastern Italy, with the basic goal of (1) assessing the planform features of paleo-channels over the last 1000 years; (2) reconstructing course changes during channelization; and (3) assessing how channel morphology has evolved over the explored timeframe. Betz et al., (2020) investigated Kyrgyzstan's Naryn River and demonstrated the ability of the Shuttle Radar Topographic Mission (SRTM)-1 elevation data and Landsat8 OLI imagery in the automatic mapping of different landscape criteria, such as riparian areas, riparian foliage growth, active channel breadth and containment, and also stream capacity. Using the InVEST (Integrated Valuation of Environmental Resources and Trade-offs) habitat quality model, Hack et al. (2020) assessed the individual and cumulative effects of built-up areas, first- and second-order road and stream pollution from urban runoff, and wastewater discharge on habitat quality across a 200 m wide river corridor. In the Panjkora River Basin (PRB), Pakistan's eastern Hindu Kush, Ullah and Zhang (2020) found flood-prone areas. 154 flood sites were reported after a comprehensive field study and analysis of Landsat-7 and Google Earth photos during the 2010 floods. The flood-prone areas in the PRB region were mapped using eight flood parameters: slope, elevation, land usage, Normalized Difference Vegetation Index (NDVI), topographic wetness index (TWI), drainage rate, and rainfall. The statistical study of the role of riparian vegetation in controlling and protecting river ecological status is the subject of Tormos et al. (2011). It reflects on the value of land use metrics and the geographic footprint over which they are measured for evaluating and executing effective and practical riparian corridor retention and conservation strategies. It looked at (1) the ability of Very High Spatial Resolution (VHRS, metric) satellite data for large scale mapping of the Riparian Area Ground Cover (RALC); (2) the application of GIS tools to produce riparian spatial measurements; and (3) the strategy of regionalized burdens.

3 Remote Sensing Platform and River Corridor Mapping

Field-based approaches of fluvial geomorphology are increasingly using terrestrial remote sensing to assess the surface morphology and measure the discharges of water, sediment, or biomass flowing through a river section. When TLS is combined with mobile platforms, it is possible to cover several kilometers of non-wetted land in large river courses at scales ranging from small gravel pads to narrow flow stretches of several hundred meters(Williams et al. 2014). Recently, drones fitted with real-time kinematic (RTK) GPS have become more widely available, accounting for centimeter-level precision in image positioning. This advancement in technology could make it much easier to use UAVs for geomorphological research. LiDAR data, multi-spectral, and hyper-spectral satellite sensors, and even RFID tracking technologies are now available on drones (Cassel et al. 2019). After the advent of multispectral satellite imagery, satellites have offered access to multiple data gathered from electromagnetic radiation that is supplementary to field-based data or aerial photos, often for vast networks (e.g., since 1970s for the Landsat series)(Henshaw et al. 2013). At large scales, Landsat 7 and 8 with images at 30 m and 15 mm resolution and a 16-day revisit capability are often used, for example, to characterize thermal patterns (Wawrzyniak et al. 2016) or channel morphology (Mandal et al. 2018). Geomorphologists have called for higher spatial resolution in recent decades, but certain geomorphic concerns can now be answered where resolution is limited (e.g., river bathymetry from radiometric resolution). Satellites have improved their spatial resolution (ranging sub-meter scales) and revisit capacity (subweekly acquisition) in recent years, accumulating multispectral and microwave data, as well as DEM datasets (e.g., SRTM, ASTER (Advanced Space Thermal Emission Radiometer)) for topographic mapping. We are approaching an age in which satellites can monitor and classify river channel planforms and functions almost weekly for major rivers all over the world. To be fully realized, this potential needs specialized and interdisciplinary experience, in addition to accessibility to capital/resources.

4 Geospatial Methods and River Corridor Mapping

Most of the approaches used, such as unmanned aerial vehicle imagery (Westoby et al. 2012), terrestrial laser scanner (Telling et al. 2017), airborne laser scanning (ALS) (Hofle and Rutzinger 2011), acoustic Doppler current profiler (ADCP) (Muste et al. 2004), and multibeam echo sounder (MBES) (Muste et al. 2004), have been extensively evaluated and may be used to notify researchers for implementation and processing (Jha et al. 2013). Due to the high point density, TLS has mainly been used to explore fine-scale texture, such as in investigating gravel bars (Heritage and Milan 2009), differences in surface texture preflood and post-flood (Picco et al. 2013), roughness across various climatic drivers (Storz-Peretz et al. 2016), and bank skin drag coefficients (Leyland et al. 2015). Roughness assessments in flume tests (Morgan et al. 2017), field studies (Piton et al. 2018), and river preservation research have both used structure from motion (SfM) strategies (Marteau et al. 2017). The flow dynamics are studied using both acoustic doppler velocimeters (ADVs) and acoustic doppler current profiling (ADCPs). The former is primarily used to analyze flow behavior like momentum and acceleration in both flume (Abad & Garcia 2006), and field environments (Wilcox and Wohl 2007). ADVs have also been used to study realistic management issues like weir construction (Bhuiyan et al. 2017) and the impact of ship wakes on near-bank traffic (Fleit et al. 2016). ADCPs have been employed at the reach continuum to evaluate flow discrepancy through changing physical structures (Guerrero and Lamberti 2011), flow connection with dune bed geomorphology (Parsons et al. 2005), and flow forms through a diverse array of tortuous, straight, and paleo streams (Guerrero and Lamberti 2011). After being tested in controlled and field environments (Simmons et al. 2017), the acoustic backscatter from MBES sensors can be utilized to estimate suspended sediment concentrations (SSC), offering the ability to obtain SSC data through feature and reach scales.SSC has been studied using medium resolution imaging (20–30 m) at the conglomeration of the Mississippi and Missouri Rivers(Umar et al. 2018), which have distinct sediment regimes, and also along the Yangtze River (Wang et al. 2011). The variety of researchers, however, prefer to utilize coarser (250 m) MODIS data, focusing on lengthy and broad rivers including the Yangtze (Wang and Lu 2010), the Amazon (Santos et al. 2018), the Changjiang (Lu et al. 2010), and the Solimoes (Espinoza-Villar et al. 2018), and relying on statistical correlations between experimental SSC values with visible and infrared bands of electromagnetic spectrum. Stanford University and Minnesota University have been developed InVEST (Integrated Valuation of Environmental Services and Trade-offs) model to measure the effect of emblematic urban intimidations on river corridor habitat eminence (Natural Capital Project Habitat Quality 2017). While the model provides for highly distinct caricatures of habitat suitability and vulnerability effects, the resultant habitat eminence, which is described as an environmental value score between 0 and 1, is quite vague and ambiguous. Thoms et al. (2018) used a series of multivariate analyses, known as functional process zones (FPZs), which are large sections of the river network that have identical hydrogeomorphological characteristics. FPZs are described and labeled using statistics derived from the river network of the drainage network and neighboring valley, as well as geological and precipitation data. Azimah et al. (2019) established a flood evacuation center along Batu Pahat's Sembrong River using the Hydraulic Model HEC-RAS in conjunction with a GIS. To assess flood-prone areas, HEC-RAS was used to generate a two-dimensional fluid dynamic analysis output.

5 Integration Machine Learning Approach and River Corridor Mapping

Maps can be created for major river systems to explore improvements in river channel width, planform structure, and the size and layout of channel and floodplain elements. Machine learning techniques have the ability to simplify the process of image and feature detection, vastly increasing the usefulness of these objects for geomorphological research. On the Colorado River in Grand Canyon, Arizona, Grams and Buscombe (2018) introduced two applications of machine-learning techniques to landform classification and measurement of landform transition from aerial and oblique images. Ren et al. (2020) used machine learning (ML) techniques (Random Forest) to categorize and map the distribution patterns of riverbed sediment grain size, as well as filling in dissimulate of substrate detail, along the reach. For the prophecy of the closure likelihood of Da Dien estuary river mouths, Anh et al. (2019) applied and evaluated various machine learning practices (logistic regression, neutron network, Bagging, AdaBoost, Random SubSpace, and LogitBoost). Casado et al. (2015) proposed a UAV-based method for identifying hydro morphological characteristics from high-resolution multispectral imagery using an innovative classification strategy based on artificial neural networks (ANNs). The system was created for a 1.4-km stretch of the River Dee in Wales, UK. Various machine learning-driven applications are being developed to simplify data exploration and close the gap in data ingestion across spatial and temporal scales. Modern high-performance computing devices can now handle huge volumes of data quickly—the world's most efficient supercomputer, for example, can process 200 petaflops of data every second.

6 Future Prospect and Conclusion

Current evidence, software, and geographical analyses are opening up cost-effective and exciting prospects to apprise river management around the world. One of the most significant scientific contests in remote sensing is to expand the size and geographic exposure at which a comprehensive and high-resolution simulation of the Earth's surface can be obtained. Looking forward, the precision of Earth observation data from satellites has improved to the point that it can quickly include more detail to illustrate big to mid-sized river topographies and improvements in space and time. Crochemore et al. (2019) examine the accuracy of 21,586 river flow time series collected from 13 publicly available hydrological databases. These repositories have been used to evaluate global river catchment parameters in recent global databases including the Global Streamflow Indices and Metadata Archive (Do et al. 2018). With the emergence of fine-scale DEMs such as MERIT (Yamazaki et al. 2017) or the medium resolution (90 m) TanDEM-X, topographic fingerprints derived from DEM (e.g., Amatulli et al. 2018) can now be considered to deliver a more comprehensive evaluation of the geographical dispersal of various river groups (Archer et al. 2018). Analytical fluvial geomorphologists are increasingly developing and using toolboxes to understand and quantify river landscape change. The open-source LSDTopoTools software is used for terrain mapping, stream network processing, chi analysis, corrosion rate estimation, rooftop flow navigation, and relief parameters, and/or topographic characteristics of floodplains and terraces (Golly and Turowski 2017). The CASCADE toolbox (Tangi et al. 2019) allows for network-scale measurement of sediment connections and monitoring of multiple infrastructure portfolios' impacts. The River Analysis and Mapping engine (RivaMap) was created to make fine-scale hydrographic datasets (such as retrieving the river center line and width) from Landsat data easier to compute in a short amount of time (Isikdogan et al. 2017). The Valley Bottom Extraction Tool (V-BET; Gilbert et al. 2016) and the Valley Bottom Confinement Tool (VBCT; O'Brien et al. 2019) are two tools that can be used to categorize channel compression categories and degrees across networks. Correspondingly, the Human–Environment Observatories network, which fetches together thirteen French and foreign observatories, particularly river observatories, has created a spatial data infrastructure (SDI). Scientists and stakeholders will use the Web GIS, metadata, and other visualization resources built in this SDI. These advancements will continue to help managers and policymakers to make smarter decisions about the complex balance of river corridors, which will support resident ecosystems and ecological services. From assessing the feasibility of river and groundwater management activities to clarifying regulatory authority under the Clean Water Act, there is a pressing need for improved hydrogeomorphic knowledge and modeling of hydrologic exchange processes.

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Chapter 15
Microplastic Pollution in Freshwater Systems: A Potential Environmental Threat

Vicky Singh and Sukalyan Chakraborty

Abstract Plastic particles of size <5 mm are called microplastics. In terms of quality, directly or indirectly, microplastics are among the contaminants of emerging concern that adversely impact freshwater systems. Such fragments of microplastic are so frangible that they could not be facilely abstracted. Most of the studies show the presence of microplastics in the ocean or sea or estuaries, and very few studies have reported the presence of microplastics in freshwater systems or inland waters such as lakes, rivers, ponds, and reservoirs. Due to high assiduousness and toxicity, these microplastic particles make the freshwater systems more vulnerably susceptible to water quality degradation and ultimately lead to microplastic pollution. There are numerous risks associated with microplastic pollution on sundry components of an ecosystem, such as plants, animals, and especially human beings. These minute particles are an earnest threat to the environment and can act as a medium to convey various toxic chemicals, metals, pathogens, antibiotics, inimical algal blooms, and can be carcinogenic to humans. Till now, there is no defined remedial technique for the abstraction of microplastics from freshwater other than filtration, but there are studies reported which can degrade these particles photo-catalytically, that too for some designated microplastics types. Some studies have reported the biotechnological techniques to degrade some designated microplastics from wastewater. This book chapter is an endeavor to define the microplastic pollution in freshwater systems and present a piece of state-of-the-art information about microplastics, risks associated, and possible remediation techniques.

Keywords Micro-plastics · Morphology · FT-IR · Freshwater systems · Water quality degradation · Remediation techniques

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1 Introduction

Plastics are a versatile gift to modern mankind. The term "plastic" was derived from the ancient Greek word "*plastikos"* and the Latin word "*plasticus"* which is attributed to substances that can be easily molded (Crawford and Quinn 2017a). Plastics are synthetic polymers derived from the polymerization of individual monomers. The first polymer developed was natural rubber latex in the year 1839 by Charles Goodyear (Lambert and Wagner 2018). The characteristics of plastics, which makes them essential material in our daily life are plasticity, resistance to chemicals, buoyancy, high durability, lightweight, and cost-effectiveness (Shim et al. 2018). This leads to the global production of plastics in an enormous amount from 1.7×10^6 tons in 1950 to 3.2×10^8 tons in 2015 (Plastic Europe 2016). Although plastics can be persistent in the aquatic environment, it can degrade and results in enormous micro- and nano-sized particles (Koelman et al. 2015).

Studies conducted on surface water and sediments of freshwater systems including estuaries (Zbyszewski and Corcoran 2011), rivers (Mani et al. 2015), lakes (Biginagwa et al. 2016), and reservoirs (Zhang et al. 2015) reported these tiny-sized particles (<5 mm) as "microplastics," which can be quantified using a microscope. Currently, there is no defined lower size of microplastics (MPs), but according to Lee et al. 2013, microplastic particles <0.001 mm are termed as nano-plastics, whereas particles >5 mm are classified as meso-plastics, and those particles of size >25 mm are macro-plastics. Based on their origin, MPs are broadly classified into primary MPs and secondary MPs. Plastics originally manufactured of size <5 mm are termed as primary MPs, whereas plastics derived from the disintegration of any external force are called secondary MPs (Sruthy and Ramasamy 2017).

Since numerous studies were inclined toward marine ecosystems, freshwater systems were given less attention. However, studies based on the ecotoxicological impact of MPs on the environment were emphasized from the last few years (Ogonowski et al. 2016; Booth et al. 2016). There is some evidence that these MPs can impact aquatic organisms of freshwater systems (Scherer et al. 2017). Ingestion of MPs by some aquatic organisms may lead to implications on environmental health (Wright et al. 2013). These MPs can be a vector to many persistent organic pollutants and can contaminate the food chain (Li et al. 2015). Besides particles toxicity, due to ingestion of MPs, these can cause additional toxic effects on the gastrointestinal tract of various species (Rochman et al. 2013). Basically, there is a knowledge gap between exposure of MPs in the environment and their effects on the ecosystem including humans.

This chapter outlines the risk associated with MPs exposure in freshwater systems. The first segment provides certain aspects of plastics and MPs. The consecutive section discusses the protocols for monitoring MPs, fate, distribution, and remedial techniques for microplastic pollution.

2 Plastic Production and its Applications

Plastics are processed with a variety of chemical additives to modify their basic characteristics and make the application for our use. In the modern age, plastics are considered as one of the essential materials for human beings. This resulted in a rapid increase in the demand and supply of these materials globally. China (26%), Europe (20%), and North America (19%) were the top three global manufacturers of plastics by the year 2014 (Plastic Europe 2015). In Europe itself, countries such as Germany (24.6%), Italy (14%), and France (9.6%) contribute approximately 60% of the total demand of Europe. Plastics have a variety of applications in almost every sector including packaging, building and construction, automotive, agriculture, household, electrical and electronics, sports and leisure, and medical equipment. Most of the plastics are used for packaging purposes. Plastics are broadly classified as thermosets and thermoplastics (Plastic Europe 2018). Thermosets are a family of plastics that cannot be reshaped and re-formed. Polyurethane (PU), Epoxy resins, Vinyl ester, Silicone, Acrylic resins, and so on are thermosets. Thermoplastics are a family of plastics that can be reshaped and re-formed. Thermoplastics include polyethylene (PE), polypropylene (PP), polycarbonate (PC), polyvinyl chloride (PVC), polymethyl methacrylate (PMMA), polystyrene (PS), polyethylene terephthalate (PET), acro-nitrile butadiene styrene (ABS), and so on. PP, PE, PET, and PS are some of the most consumed plastics across the globe (Plastic Europe 2018). It was forecasted that by 2050, plastic production will vary between 850 million tons/year (Shen 2009) to 1124 million tons/year (Neufeld et al. 2016).

3 Microplastic Pollution in Inland Waters

MPs are considered surface water pollutants due to their density. Most of the plastic materials are denser than water, which compels the plastics to float. This resulted in the accumulation of immense plastic garbage in five main gyres: The North Atlantic Gyre, The South Atlantic Gyre, The North Pacific Gyre, The South Pacific Gyre, and The Indian Ocean Gyre (Lebraton et al. 2012). The garbage trapped in the swirling vortex of the North Pacific Gyre is known as The North Pacific Garbage Patch and is a serious environmental challenge (Mitchell 2015). It was estimated that approximately 25,000 pieces of plastics per km^2 are floating in the North Atlantic Gyre and approximately 20,000 pieces of plastics per km^2 are floating in the South Atlantic Gyre (Moore et al. 2001; Law et al. 2010). The Mediterranean Sea is considered one of the most polluted regions of the world and is polluted due to plastic materials (Crise et al. 2015). Thus, plastic pollution is clearly evident in the oceans.

In the case of freshwater systems, plastic pollution may be the outcome of direct littering of plastic debris, leisure or recreational activities, drainage systems, flooding, storms, or transportation of plastics by urban water sources (Crawford and Quinn 2017b). It is a fact that MPs in freshwater systems are less acknowledged than MPs in marine ecosystems (Wagner et al. 2014). Numerous publications reported the presence of MPs in marine ecosystems which includes poles (Obbard et al. 2014), tropics (Costa and Barletta 2015), pelagic open ocean habitat (Kooi et al. 2016), deep-sea benthic habitat (Taylor et al. 2016), deep-sea corals (Woodall et al. 2014), and coastlines of continents (Browne et al. 2011).

In contrast, publications based on MPs in freshwater systems have given attention in the last two decades. The first study of MPs in inland waters was reported on North American rivers by Haze and Cormons in 1974. Then more emphasis was given to freshwater systems, especially lakes. Some of the studies reported on freshwater systems are summarized in Table 15.1.

MPs can immigrate into freshwater systems through different pathways. Land-based activities, as well as water-based activities, contribute to MPs concentration in freshwater systems (Njeru 2006; Pon and Becherucci 2012). Land-based activities include direct littering of plastics, agricultural activities (Xu et al. 2018), wear and tear of tires (Councell et al. 2004), constructional activities, discharge of wastewater (Lambert and Wagner 2018), washing of clothes, and usage of personal care products. Water-based activities include shipping, fishing, and recreation activities (Sruthy and Ramasamy 2017). MPs can be transported from land to water via stormwater (Lambert et al. 2014). The movement of MPs depends on factors such as weather conditions, land cover type, the distance of freshwater system from land, type of plastic waste, and soil characteristics (Lambert and Wagner 2018).

3.1 Morphological Characterization of Microplastics

MPs are derived from plastic materials. There are numerous types of plastics with specific characteristics. This implies that MPs, which are originally originating from plastics, will also have different characteristics. These different characteristics of MPs were differentiated into morphological characterization and chemical characterization. Morphological characterization of MPs includes shape, size, and color.

3.1.1 MPs Shape

MPs are classified as fragments, fibers, films, foams, microbeads, and pellets in terms of shape (Sruthy and Ramasamy 2017). However, there is no well-defined protocol for the identification of the shape of MPs in freshwater systems. The shape of these MPs can be used for source determination and trace their pathways. For example, a study conducted by Peng et al. (2018) in sediments of the freshwater river of Shanghai, China, reported that sphere-type MPs were maximum (88.98%) followed by fibers (7.55%), and fragments (3.47%). These MPs were generated from land-based sources such as cloth washing, upstream runoff, shipping, or waste dumping. Source identification of MPs in Vembanad lake, Kerala, was carried out by Sruthy and Ramasamy (2017) in which she reported that the probable source of

Table 15.1 A summary of MPs pollution in various freshwater systems across the globe

Freshwater system			
Lakes	Location	Abundance	References
Lake Geneva	Switzerland	4.81×10^4 p/km^2	Alencastro (2012)
Wuhan Lakes	China	8.93×10^3 p/m^3	Wang et al. (2017)
Taihu lake	China	6.8×10^6 p/km^2	Su et al. (2016)
Taihu lake	China	2.58×10^4 p/m^3	Su et al. (2016)
Lake Hovsgol	Mongolia	4.44×10^4 p/km^2	Free et al. (2014)
Lake Winnipeg	Canada	7.48×10^5 p/km^2	Anderson et al. (2017)
29 great lakes tributaries	USA	32 p/m^3	Baldwin et al. (2016)
Laurentian great lakes	USA	4.66×10^5 p/km^2	Eriksen et al. (2013)
Lake Huron	Canada	3209/85 m^2	Zbyszewski and Corcoran (2011)
Vembanad lake	India	252.8 ± 25.76 p/m^2	Sruthy and Ramasamy (2017)
Lake Garda	Italy	1108 ± 983 p/m^2	Imhof et al. (2013)
Remote lakes in Tibet	China	8–563 p/m^2	Zhang et al. (2016)
Dongting and Hong lakes	China	900–4650 p/m^3	Wang et al. (2017)
Lake Superior	USA	0–110 p/1000 m^2	Hendrickson et al. (2018)
Lake Michigan	USA	0–100 p/1000 m^2	Mason et al. (2016)
Rivers			
Danube river	Austria	141.7 p/m^3	Lechner et al. (2014)
Rhine river	Europe	3.9×10^6 p/km^2	Mani et al. (2015)
Dutch river delta and Amsterdam canal	Netherlands	1.87×10^5 p/m^3	Leslie et al. (2017)
Los Angeles river, San Gabriel river, coyote creek	USA	1.29×10^4 p/m^3	Moore et al. (2011)
Raritan river	USA	–	Estahbanati and Fahrenfeld (2016)
Saigon river	Vietnam	172–519 p/L	Lahens et al. (2018)
Seine river	France	182–200 p/m^3	Dris et al. (2015)
Raritan river	USA	1.3–3.5 p/m^3	Estahbanati and Fahrenfeld (2016)
Estuaries			
Yangtze estuary	China	1.02×10^4 p/m^3	Zhao et al. (2014)
Goiana estuary	Brazil	0.19 p/m^3	Lima et al. (2014)
Changjiang estuary	China	23.1 ± 18.2 n/100 L	Xu et al. (2018)
Tamar estuary	UK	–	Sadri and Thompson (2014)
Pearl river estuary	Hong Kong	5595 ± 27.417p/m^2	Fok and Cheung (2015)
5 urban estuaries of KwaZulu-Natal	South Aftrica	745.4 ± 129.7 p/500 mL	Naidoo et al. (2015)
4 estuarine rivers	USA	< 1.0 to >560 g/km^2	Yonkos et al. (2014)
Gulf of Mexico estuary	USA	5–117 p/m^2	Wessel et al. (2016)

(continued)

Table 15.1 (continued)

Freshwater system Lakes	Location	Abundance	References
Reservoirs			
Three gorges dam	China	1.36×10^7 p/km^2	Zhang et al. (2015)
Three gorges dam	China	1.26×10^4 p/km^3	Di and Wang (2018)
Teltow Canal	Germany	0.01–95.8 p/L	Schmidt et al. (2018)

films was the breakdown of plastic carry bags, fibers from fishing nets, foams from packaging materials, and microbeads from personal care products. In the Three Gorges Reservoir, fragments and films were the most dominant type of MPs (Zhang et al. 2015), whereas fibers, granules, films, and pellets were predominant in the freshwater of Wuhan (Wang et al. 2017).

3.1.2 MPs Size

As the size of MPs decreases, detection of these MPs also gets more difficult. A study conducted on lake Hovsgol by Free et al. (2014) reveals that 40% of MPs fall into the size class of 0.355 mm–0.999 mm and 40% of MPs fall into the size class of 1.00 mm–4.749 mm. Size distribution of MPs observed by Peng et al. (2018) was 31.19% of total MPs were < 100 μm, 3.56% of total MPs were between 500 μm and 1000 μm, 2.8% of total MPs were between 1000 μm and 5000 μm, and most of the MPs (62.15%) lies between 100 μm and 500 μm. In Yangtze Estuary, Zhao et al. (2014) classified MPs into four size categories, that is, 0.5 mm–1 mm (67%), 1 mm–2.5 mm (28.4), 2.5 mm–5 mm (4.4) and > 5 mm (0.2%). The size pattern of MPs can be related to the degree of weathering and origin of MPs.

3.1.3 MPs Color

Color of MPs was inherent from their parent plastics, but MPs can get discolored due to continuous weathering. The color of MPs can be used to indicate the potential to be ingested by various aquatic organisms (Lambert and Wagner 2016). In Taihu lake, the color of detected MPs was blue, black, green, yellow, white, and transparent among which blue color was the dominant one (Su et al. 2016). Similar study was done by Peng et al. (2018) in which he observed that most of the MPs were white (90%). Other colors of MPs observed includes blue (3%), transparent (3%), white (2%), red (2%), and transparent. Further investigation on the impact of MPs color on ecology and environment needs more attention.

3.2 Chemical Characterization of MPs

Chemical characterization of MPs is done in terms of the type of polymer. A variety of polymers were used for the manufacturing of plastic products. Different polymers possess different unique properties which include strength, elasticity, persistence, and buoyancy. In case of freshwater systems, generally FT-IR technique (Peng et al. 2018) and Raman spectroscopy (Sruthy and Ramasamy 2017) were adopted for the identification of polymer type. μ-FT-IR technique was used for identification of polymer type by Peng et al. (2018) in which he identifies that polypropylene contributes 51.1%, polyester constitutes 17.1%, rayon constitutes 11.4%, cotton constitutes 5.7%, phenoxy resin constitutes 2.9%, and poly(vinyl stearate) constitutes 2.9%. Raman spectroscopic technique was adopted by Sruthy and Ramasamy (2017) reveals that low-density polyethylene was most abundant, followed by polystyrene and polypropylene. Low-density polymers such as polyethylene, polypropylene, and polystyrene were identified in Three Gorges Reservoirs (Zhang et al. 2015). Transparent cellophane was detected in Tahiu lake (Su et al. 2016).

4 Fate and Impact of MPs

Once the MPs are exposed to freshwater systems, it poses a risk to the environment in various ways. MPs having different morphological and chemical characteristics encompass several risks to biota (Eerkes-Medrano and Thompson 2018). Studies reported that MPs reaching freshwater systems may undergo photochemical degradation, thermal degradation, or biological degradation (Lambert et al. 2013). The MPs can degrade by a single mechanism or incorporation with other mechanisms (Lambert and Wagner 2016). Degradation mechanisms gravitate to investigate the tensile strength, weight loss, depolymerization of molecular structure, and microbial strain of specific type of MPs (Gigault et al. 2016). The breaking down of MPs further leads to lower size fractions and even nano-size fractions.

As soon as these MPs disintegrate from parent material, these particles are exposed to the biota of the freshwater system. These MPs can be ingested by a variety of organisms and enters into the food chain. Now, MPs can directly impact the organism, or it can be transferred to higher trophic levels. For instance, if MPs are ingested by fish in large amounts, fish will die due to starvation, whereas, while MPs are ingested in small amounts by fish, it may be consumed by humans and can impact indirectly (Bem 1990).

4.1 Impact on the Environment

MPs possess a ubiquitous environmental threat to flora and fauna (Prata et al. 2019). The environmental impacts of MPs are categorized as biological, chemical, and physical. The biological impacts include the transfer of microorganisms that colonize on the surface of MPs (Oberbeckmann 2015). As the size of MPs decreases, the surface area to volume ratio increases, which allows enormous microbes to adhere to the surface of these tiny particles (Crawford and Quinn 2017c). MPs have the potential to support the growth of microbial colonies on their surface. Limited literature are available on the introduction of pathogens attached with MPs. A study conducted by McCormick et al. (2014) on an urban river in Chicago, Illinois, reveals that some microbes and pathogens were attached to the surface of MPs which degrades the plastic materials. This study also suggested that these attached microbes and pathogens can get transported from freshwater systems to higher trophic levels. A study conducted on Yangtze Estuary, China, confirms that potential pathogens were attached to these suspected MPs (Jiang et al. 2018). Morphological characteristics get tampered which includes increment of the density of MPs and reduction of hydrophobic nature of MPs, due to biofilms attached on the surface of MPs (Carr et al. 2016).

Ingestion and entanglement of MPs by the organisms of freshwater systems are the main physical impact of MPs on the environment (Li et al. 2018). More than 200 species get affected due to entanglement and ingestion of plastic debris (Laist 1997). These MPs look like food to the aquatic species. Once ingested, they may die due to starvation or suffocation (Derraik 2002). Exposure to MPs results in reduced body size and reproduction efficiency in *Daphnia magna* (Besseling et al. 2014). Exposure to polystyrene-type MPs resulted in chronic effects to larvae of the brine shrimp *Artemia franciscana* (Bergami et al. 2016). Della Torre et al. (2014) reported that developmental defects were observed in the genes of urchin embryos(*Paracentrotuslividus*) after fertilization. Cole et al. (2015) reported that the reproductive capability of pelagic copepod, *Calanus helgolandicus,* gets affected after MPs ingestion. Once ingested, MPs can get transferred from digestive systems to adjacent tissues (Harmon 2018). MPs can translocate from the gut of the mussels (*M. edulis)* to the circulatory system (Browne et al. 2011) or MPs from gills of *M. edulis* reaches stomach, transferred to digestive glands, and ultimately gets accumulated in lysosomal systems, resulting in inflammation (von Moos et al. 2012). MPs can get transferred from the stomach to the liver of fish (Avio et al. 2015). Ingestion of MPs by lower trophic levels such as fish (Rochman et al. 2013) gets transferred to higher trophic levels such as birds (Romeo et al. 2015), turtles (Bugoni et al. 2001), dolphins (Denuncio et al. 2011), or humans (Mattsson et al. 2015). Bioaccumulation of MPs to higher tropic levels is a serious concern since humans are ultimate consumers.

Chemical impact includes toxicity to living organisms through various pathways and mechanisms (Li et al. 2018). Various chemicals such as additives, heavy metals, colorants, plasticizers, and stabilizers are used for the production of plastic polymers (Murphy 2001). Sussarellu et al. (2016) conducted a study on oysters by exposing MPs for 2 months and reported that oocyte number, diameter, and sperm velocity get decreased. Additionally, heavy metals present in MPs get desorbed into freshwater systems and can enter the food chain (Li et al. 2018). Chemicals like polybrominated diphenyl ethers (PBDEs) and hexabromocyclododecanes (HBCDs) used in the production of plastic polymers can be desorbed from MPs into freshwater systems during the degradation process (Sun et al. 2016). The chemicals get sorbed into MPs and get desorbed into the stomach of fish and can translocate through blood circulation (Carpenter et al. 1972; Chen et al. 2006). MPs can also act as a vector for hydrophobic pollutants such as polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), hexachlorocyclohexanes (HCHs), halogenated hydrocarbons, dichlorodiphenyltrichloroethane (DDT), and organochlorine pesticides which are highly toxic and persistent in nature (Velzeboer et al. 2014; Rios et al. 2010). Metal toxicity due to copper and zinc released from antifouling paint and gets adsorbed by microbeads was reported by Brennecke et al. (2016).

4.2 Impact on Humans

There is a lack of sufficient evidence of MPs ingestion by humans. However, a study reported the transition of MPs through the gastrointestinal tract in significant volume (Tomlin and Read 1988). A similar study was conducted on human volunteers in the late 1990s, in which the transition of MPs through the gastrointestinal system was observed. Incidentally, no confirmed effect on humans is reported by the author, but certainly, the reduction of nutrients decreases, and nutritional deficit can occur eventually (Erdman et al. 2012). At the same time, MPs contain various additives and toxic chemicals that can impact humans in various ways (Crawford and Quinn 2017c).

Studies confirmed that MPs can act as a vector to waterborne contaminants in humans. The most common pollutant that can contaminate MPs is polycyclic aromatic hydrocarbons (PAHs) (Frias et al. 2010). The contaminated MPs reach human body through fish (Dhananjayan and Muralidharan 2012). Several studies reported the presence of PAHs in fish such as *Eleutheronematetradactylum, Sardinella longiceps, Otolithesruber, Mystusseenghala, and Coiliadussumieri*(Dhananjayan and Muralidharan 2012). Other pollutants detected in fish are polybromide diphenyl ethers (PBDEs) (Liu et al. 2011) and polycyclic biphenyls (PCBs) (Van Cauwenberghe et al. 2015). Potential exposure of these carcinogenic pollutants along with MPs to humans can be dangerous. Continuous monitoring of the exposure level and impact on humans is required.

5 Remediation Techniques

Remediation of microplastic pollution can be categorized into three categories, that is, legislation, technological tools, and biotechnological tools (Pico et al. 2019). To control the MPs pollution, countries such as the USA enacted a law, namely, "*Microbead-Free Water Act*", in December 2015. A regulation has been proposed by EU packaging and use of lightweight carry bags. In developing countries such as India, government bans the single-use plastics in 2019. In addition, engineering tools such as water treatment plants (WTPs) should be given more emphasis to remove MPs from drinking water. A study conducted for the removal of MPs from wastewater reported the removal efficiency of up to 95% from effluents (Talvitie et al. 2017). Nanotechnological advancement such as visible light photocatalytic degradation was adopted to degrade LDPE residues with zinc oxide nanorods (Tofa et al. 2019).

The most obvious mitigation technique is the identification of the source and sink of MPs. The expected source of MPs can be production, commerce, consumer, and mismanagement of waste (Lambert and Wagner 2018). A circular economy of plastic waste with the support of stakeholders should be initiated, which includes green chemistry, production and use of biodegradable polymers, the responsibility of producers, and awareness in terms of reusing, reducing, and reducing use of plastics.

6 Conclusion

MPs pollution in freshwater systems is an emerging concern. In this chapter, a brief introduction of MPs pollution in freshwater systems has been provided. MPs have been detected in almost all freshwater systems globally. Continuous monitoring needs more attention. The morphological characterization reported in various studies has been done in terms of shape, size, color, and polymer type can be used to determine the origin of these MPs. The ecological and biological risk associated with MPs exposure to the environment has been discussed in this chapter. Various toxic chemicals associated with MPs can be a potential threat to the environment. However, more precise analysis needs to be done to understand the potential risk. MPs are detected in aquatic organisms and humans in a significant amount, but the impact is still not defined clearly. Remediation and control measures should be given more emphasis. Literature-based remediation techniques of MPs from freshwater systems are lacking and need more attention.

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Chapter 16
Sand and Gravel Mining and its Consequences on Morphometry of Raidak-II River in Eastern Dooars, India

Debasish Talukdar and Tapan Kumar Das

Abstract The sand and gravel mining plays a significant role in changing the morphometric features of any river like shifting of river bank, channel width, channel depth, channel pattern, etc. These impacts become evident due to unscientific sand and gravel mining which was exponentially increased nowadays. Raidak-II River, consisting 28 mining sites in its course from Bhutanghat to Binnaguri, is well known for maximum number of sand and gravel mining sites of Dooars region of West Bengal. The irrational sand and gravel mining using heavy vehicles and machineries withdraw more sediments than that are naturally offered for removal. Ultimately the health of the river has been affected adversely and these adverse impacts become evident day by day. The present study shows that most part of the left river bank has been shifted toward east for sand and gravel mining compared to the other bank causing more width depth ratio. The mid-channel bars and point bars are dissected and new small channels are formed with consequent higher braiding index. The study reveals that the long profile of the river has been affected forming irregular features with repeated aggradation and degradation. Finally, this study has attempted to assess a perception study of different contributing factors and detected that sand and gravel mining plays most profound action in changing morphometric features of this river compared to other anthropogenic activities on river.

Keywords Sand and Gravel mining · Irrational and unscientific mining · Braiding Index · Anthropogenic activities

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1 Introduction

The flow regime and sediment distribution processes of a river are closely related to temporal and spatial changes in channel geometry and hydrodynamics (Tamang 2013). From an early period of time, human beings have been using natural resources from river for various purpose according to their needs. The man and river relationship is being disturbed in many ways due to limitless extraction of river bed materials and use of natural resources by human beings. Thus, people have been changing their nearby environment which is very common nowadays (Mossa and McLean 1997). Rivers, among many gifts of nature, are now under tremendous pressure due to different forms of human involvement (Padmalal and Maya 2014). The relationship between man and river was started when the primary activities like farming and fishing were initiated. These collaborations have transformed the nature-dominated river into a human-dominated river system with tremendous negative effects of man on river conditions (Hoffmann et al. 2010). Indeed, human exercises will also modify the morphology of the channel on a large scale than natural disasters, such as flooding, droughts, landslides, and avalanches (Rinaldi 2003). In construction works, developing countries like India are still dependent on the river for their raw materials, such as, sand and gravel which can be extracted easily from the river (Kondolf 1994; Martín-Vide et al. 2010; Padmalal and Maya 2014; Rinaldi et al. 2005). Before the sand and gravel mining action, there was an overall balance between erosion and deposition. But the higher demand in the construction sector than the natural limit of replenishment leads to the destruction of the riparian environment. These impacts of sand and gravel mining can be classified into three categories: morphological, hydrological, and environmental (Rinaldi et al. 2005; Barman et al. 2017; Talukdar and Das 2020 and Bhattacharya et al. 2020). Numerous investigations have been studied on morphological effects of sand and gravel mining from the riverbed, for example, (a) upstream and downstream incision by Galay (1983), Collins and Dunne (1989), Rinaldi et al. (2005); Liangwen (2007), Martín-Vide et al. (2010). (b) Changes in channel morphometry by Surian (1999, 2006), Rinaldi (2003), Surian and Rinaldi (2003), Gob et al. (2005), Padmalal et al. (2008), Melton (2009), Ghosh et al. (2016), Singha (2017), Bhattacharya et al. (2019a, b). (c) Sediment transport and its impacts by Kondolf (1997), Rinaldi and Simon (1998), Gaillot and Piegay (1999), Sreebha (2008), Leeuw et al. (2010), Wallick et al. (2011), Tamang (2013). The sand and gravel mining plays a significant role in changing the morphology of any channels like channel width, depth, channel pattern, etc. These impacts are evident due to unscientific sand and gravel mining which was exponentially increased in European nations after 1950, but at the end of the twentieth century, the same scenario was started in Asian nations also. Sand mining is a rising activity in most of the rivers in the agricultural nations in Asia, for example, China, Malaysia, Sri Lanka, Nepal, Maldives, Vietnam, and India. The increasing demand for sand and gravel (India has the world's third-biggest constructional business after China and the USA), along with feeble governance and a higher degree of corruption, encourages uncontrolled and illegal mining of sand and gravel in the streams and coasts of India (Singh et al. 2014). Unlawful sand and gravel mining has been flourishing in different Indian states including Madhya Pradesh, Bihar, Haryana, Karnataka, Goa, Andhra Pradesh, Rajasthan, Chhattisgarh, Orissa, and West Bengal. It is likewise illicitly practiced in different states like Kerala, Tamil Nadu, Maharashtra, Gujarat, Uttar Pradesh, and Uttarakhand (Singh et al. 2014), which change the morphology of the Indian rivers. In West Bengal, random sand mining is practiced in large rivers like Damodar, Ajay, Dwarakeswar, and Mayurakshi (Ghosh et al. 2016). As per the data shown by the department of irrigation, there are 248 sand and gravel mining sites working in the state and most of the sites are dominated by sand mafias for illegal mining. Raidak-II river is one of the affected river of northern part of West Bengal where both availability of sand and the quality of the sand are better compared to the nearby areas (Talukdar and Das 2020). That is why the sand and gravel are quarried randomly from the stream bed of Raidak-II in different sites, mainly at Bhutanghat, Joydebpur, Hemaguri, Chengmari, Purba Chakchaka, Takuamari, Binnaguri, etc. resulting in unstable and ever-changing morphometric features. The existing literature shows limited works on geomorphic changes of river due to sand and gravel mining. So, the goal of this paper is to figure out the current scenario of sand mining and evaluate some geomorphic impacts of sand and gravel mining on river system. The river bank shifting, width depth ratio, spatiotemporal changes of long and cross profiles, river braiding, and short-term fluctuations of river bed morphology from the pre-monsoon to post-monsoon periods are studied and analyzed to explain the impacts of sand and gravel mining. Finally, the researcher attempted to account the liability of sand and gravel mining as a contributing factor of morphometric change of river Raidak-II compared to the others factors that are usually acting on this river system.

2 Methodology

2.1 Study Area

The Raidak river is originated from Bhutan and familiar as Wang Chhu in Bhutan. It takes entry in India at Bhutan Ghat in Alipurduar District with its name Raidak. It has been bifurcated at Tiyabarighat into two sub-streams, Raidak-I and Raidak-II. The present study focuses on the sand and gravel mining and its consequences on the lower course of eastern sub-stream of Raidak river, that is, Raidak-II from Bhutanghat to Binnaguri where it joins Sankosh river in Cooch Behar District (Fig. 16.1).

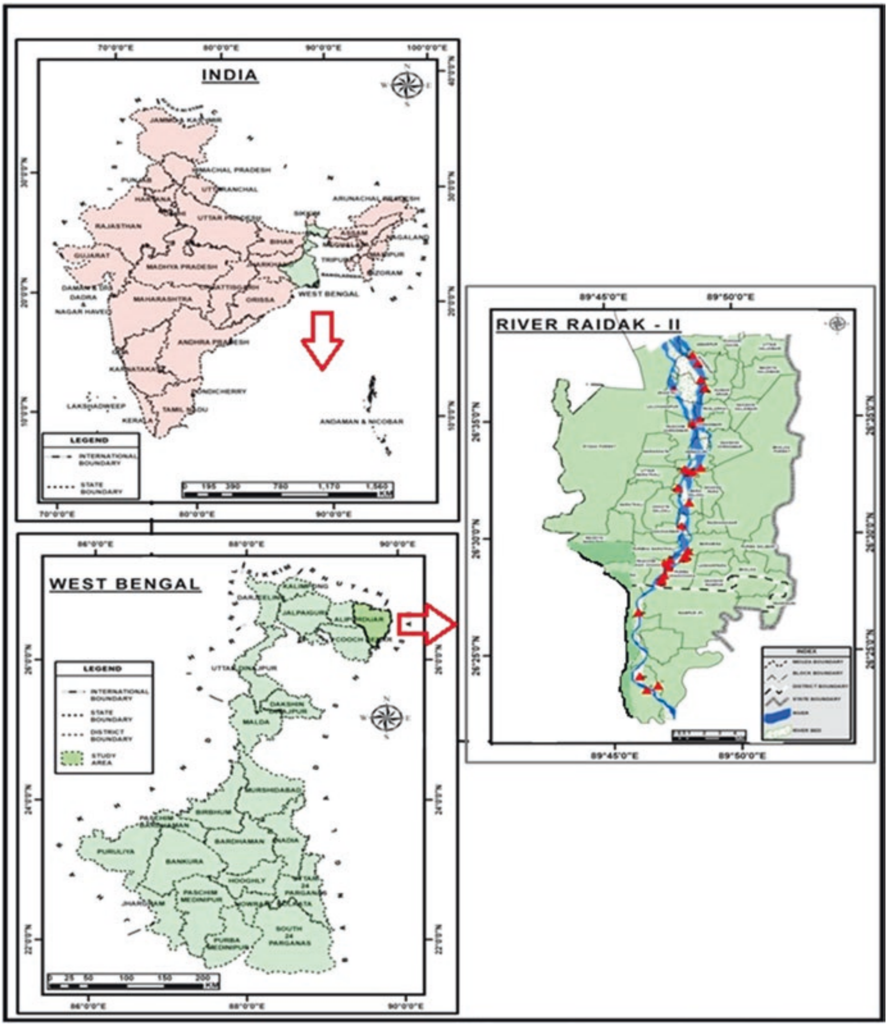

Fig. 16.1 Location map of the study area

2.2 Field Survey and Analysis of Data

Intensive field surveys with Garmin (76CSx) GPS were conducted to verify location and measure the actual areas of the mining sites with the spatial data collected from DL & LRO office Alipurduar and Cooch Behar.

The cross sections of the River-II at Bhutanghat, Purba Chakchaka, and at Binnaguri, for both pre-monsoon and post-monsoon period during three consecutive years 2018, 2019, and 2020 are drawn and analyzed in Microsoft excel and SPSS software version 20 utilizing Dumpy Level Survey data with the assistance of rise and fall method and the data collected from Central Water Commission (Jalpaiguri & Kolkata Office).

The long profiles of the Raidak-II river for the three consecutive years 2018, 2019, and 2020 are drawn using Google earth image and analyzed in Microsoft excel and SPSS software version 20.

The width depth ratio and year-wise depth of thalweg of all sites from 2018 to 2020 have been measured using the same techniques.

Impacts of sand and gravel mining on the bank shifting of the river are identified by mapping river channel dividing it into six segments (Fig. 16.2) using Google earth image of 2010, 2015, and 2020 on Arc GIS software version (10.5). Then the trend of bank line shifting has been evaluated quantitatively and qualitatively along with field verification in respect of each sand and gravel mining sites.

The Braided Index of each segment 2010, 2015, and 2020 is calculated for analyzing its temporal changes and its relation with sand and gravel mining.

Finally, the rates of contributing weightage of dominant parameters of morphometric changes, such as, discharge, gauge height, sediment load, bank materials, slope, embankment protection measures and stability, riparian vegetation, agriculture on river bed, fishing including the impact of sand and gravel mining are collected by intensive field survey and from secondary sources. The rating of these parameters are made on the basis of expertise knowledge, field experience, and community perception. Scores of each influencing factor are then plotted to visualize their contribution.

3 Results and Discussion

3.1 River Bank Shifting

The study area is consisting 28 river bed mining sites of which 17 sites are nearer to eastern bank of the Raidak river and rest of the sites are nearer to its western bank. The entire river under study is divided into six segments from Bhutanghat to Binnaguri (Segments-A to Segments-F) from upper reach to lower reach (Figs. 16.2 and 16.3). The location of mining sites, name of the lease holder/owner of the mining site, area of approved mining, production capacity in cubic feet, and per day extraction of sand from each site are shown in the following table (Table 16.1). The sharp decrease of the slope near the foothills of Himalayas causes the loss of river energy and deposition of gravels and sand resulting from the braided channel in the study area but the shifting of river bank and braiding is enhanced drastically due to the gravel and sand mining. It is observed that the highest amount of sandmining recorded in the Segment-E during 2020 amounting 3,838,071 cubic feet followed by the Segments-B, A, and D compared to the other segments. The field study confirms that the extraction of sand and gravel mining causes sudden breaks of slope and creates random depression in the river bed (Fig. 16.4a).

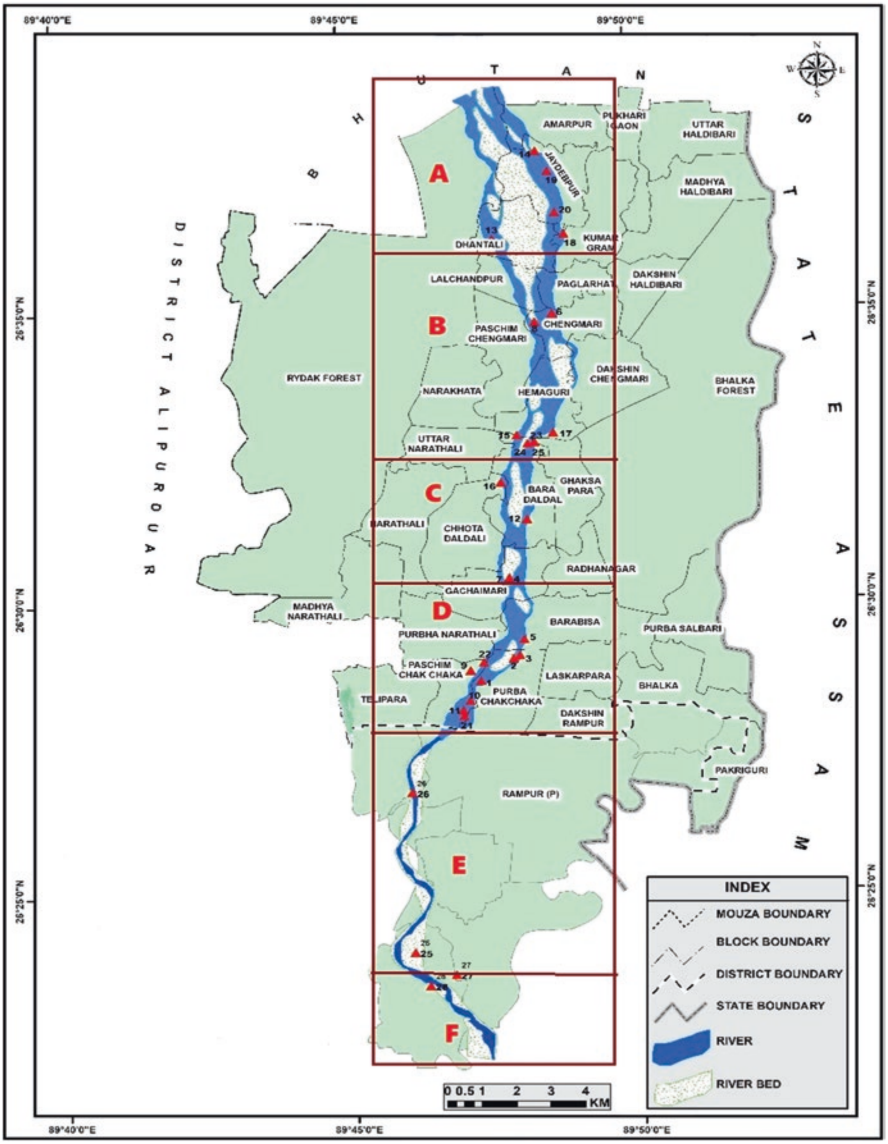

Fig. 16.2 Segments of the Raidak-II river valley

Long channel shape cutting or big circular depression consisting abrupt steep slope has been resulted due to irrational mining in most of the mining sites of eastern bank (Fig. 16.4b–d). Thus, nearer to the eastern bank at mining sites nos. 2, 3, 4, 5, 9, 10, and 18 enhanced the eastward shifting of bank due to removal of bank and bed materials as well as more concentration of water and consequent scouring specially along the concave bank. Thus, extraction of the sand and gravel mining

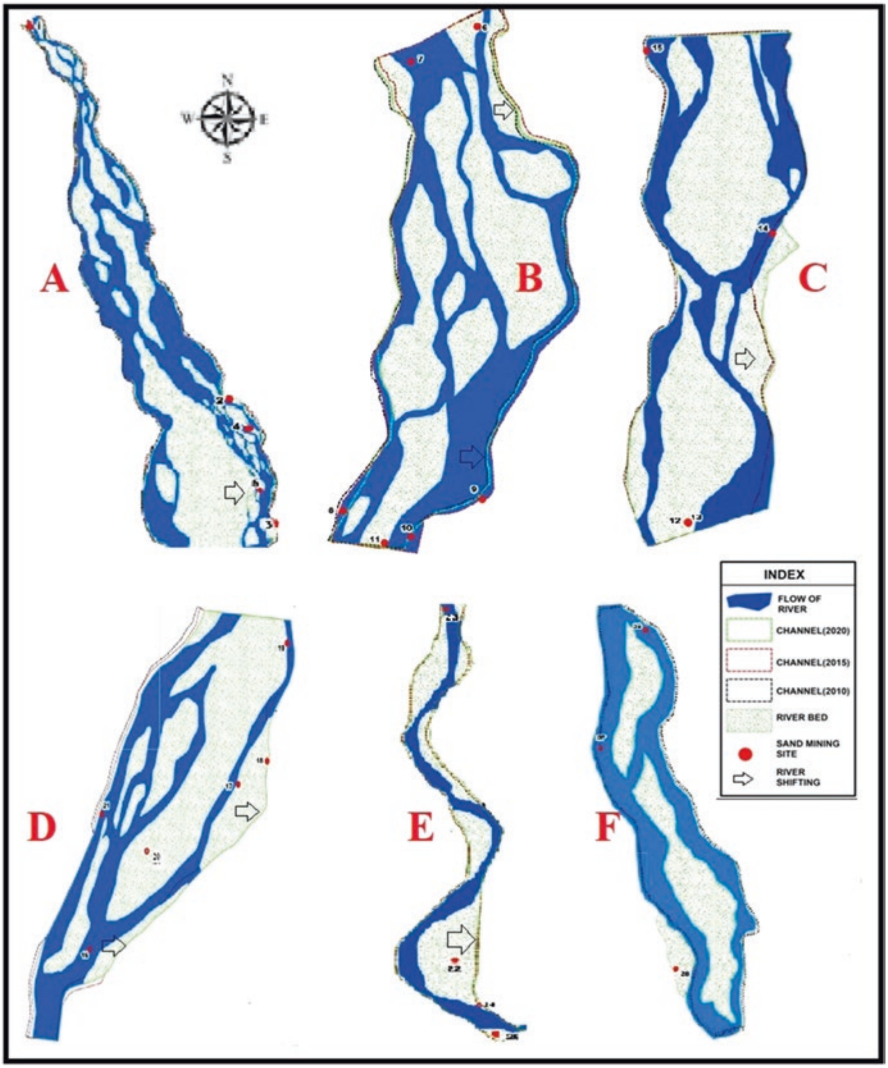

Fig. 16.3 Braiding in different segments of the river and shifting of banks

boosted the eastern oscillation of the river flow. Moreover, the random and illogical mining without compromising the natural offer of sand and gravel made the river bed more undulated and abrupt causing loss of energy and further deposition of sand and gravels in these sites. The mining site in the opposite bank, that is, the western bank of the river Raidak-II, is less fragile as it is convex in nature and sand and gravels are naturally offered for extraction and human use. Hence, the collection of materials from mining sites no-1, 7, 8, 15, 21, and 28 is more viable compared to the other mining sites (Table 16.1 and Fig. 16.3).

Table 16.1 Site wise of Sand and Gravel Mining (Source: DL & LRO, Alipurduar and Cooch Behar)

Sl. No		Name of mining site	Name of the owner	Area in acre	Yearly extraction in ft^3	Daily production in ft^3(250 days/ year)
1	*Segment-A*	Bhutanghat	Bhupesh Das	2.6	296,688	1186.752
2		Amarpur	Nityananda Das	3	341897.6	1367.5904
3		Kumargram	Rrajkumar Barman	3	341897.6	1367.5904
4		Joydevpur	Swapan ray	3	800936.64	3203.74656
5		Paglar hat	Dilip Kr Das	3	800936.64	3203.74656
6	*Segment-B*	Chengmari-i	Brijesh Kumar Sharma	8	912668.8	3650.6752
7		Chengmari-ii	Rajkumar Barman	3	341897.6	1367.5904
8		Hemaguri-i	PrafullaAdhikari	3	341897.6	1367.5904
9		Hemaguri-ii	Rajkumar Barman	3	341897.6	1367.5904
10		Borodaldali-i	Gakul Roy	3	800936.64	3203.74656
11		Barodaldali-ii	Maniram Das	3	774238.75	3096.955
12	*Segment-C*	Gachimari-i	Subhrajit Sarkar	3	341897.6	1367.5904
13		Gachimari-ii	Subhrajit Sarkar	9	1028518.4	4114.0736
14		BoroDaldali-iii	Bidyut Sarkar	3	341897.6	1367.5904
15		ChotoDaldali	Sukumar Biswas	3	341897.6	1367.5904
16	*Segment-D*	PurbaChak Chaka-i	LaxmanSaha	7.5	858982.4	3435.9296
17		PurbaChak Chaka-ii	LaxmanSaha	3	341897.6	1367.5904
18		PurbaChak Chaka-iii	Subhrajit Sarkar	3	341897.6	1367.5904
19		Barobisha	Pratap Chandra Roy	3	341897.6	1367.5904
20		Purba Chakchaka-iv	Sankar furniture &Keshab Sarkar	2.99	341897.6	1367.5904
21		Purba Nararthali	Swapan ray	3	525058.46	2100.23384
22	*Segment-E*	Takoamari	Pratap Saha	3.5	674238.75	2696.955
23		MadhurBasa	Rakib Hossain	12	1152944.2	4611.7768
24		Jalahghat	Santiram Sarkar	12	952944.2	3811.7768
25		Mahishkuchi	Santiram Sarkar	12	1057944.2	4231.7768
26	*Segment-F*	Phersabari	Tapas Dey	3.9	754489.11	3017.95644
27		Bhanukumari	Amit Mitra	2.11	690506.33	2762.02532
28		Binnaguri	Badal Das	3.5	724238.75	2896.955

3.2 Channel Braiding

Segment wise, the total sand and gravel extraction have been increased gradually from 2010 to 2015 and 2015 to 2020 (Fig. 16.5). In the mining site nos. 2, 3, 4, 5, 6, 8, 17, and 26, the random sand and gravel mining with machine specially during dry

Fig. 16.4 (**a**) Big depression in the river bed formed due to sand and gravel mining. (**b**) Long channel developed due to irrational mining. (**c**) Subsidence and recession of riverbank. (**d**) Steep slope bank due to scoured mining and collapsing. (**e**) and (**f**). Secondary channels developed due to random mining. g and h. River bed sand and gravel mining are not offered naturally by the river for human use. (**i**) and (**j**). Heavy vehicles and machineries illegally used in sand and gravel mining. (**k**) and (**l**). River bed used for fruit and vegetable production. (**m**) Vegetation protects vulnerable river bank. (**n**) River bank protected by stone wall. (**o**) Fishing net hold by bamboo causes deposition of sediments. (**p**) Bamboo bridge across the river

Fig. 16.4 (continued)

winter season made the channel bars more dissected and new small channels are formed (Fig. 16.4e–f). Thus, the braided index for each segment for the year 2010, 2015, and 2020 are gradually increasing as an impact of formation of channels within the point bars and mid-channel bars of the river due to extraction of sand and gravels creating new channels and diverting water flows.

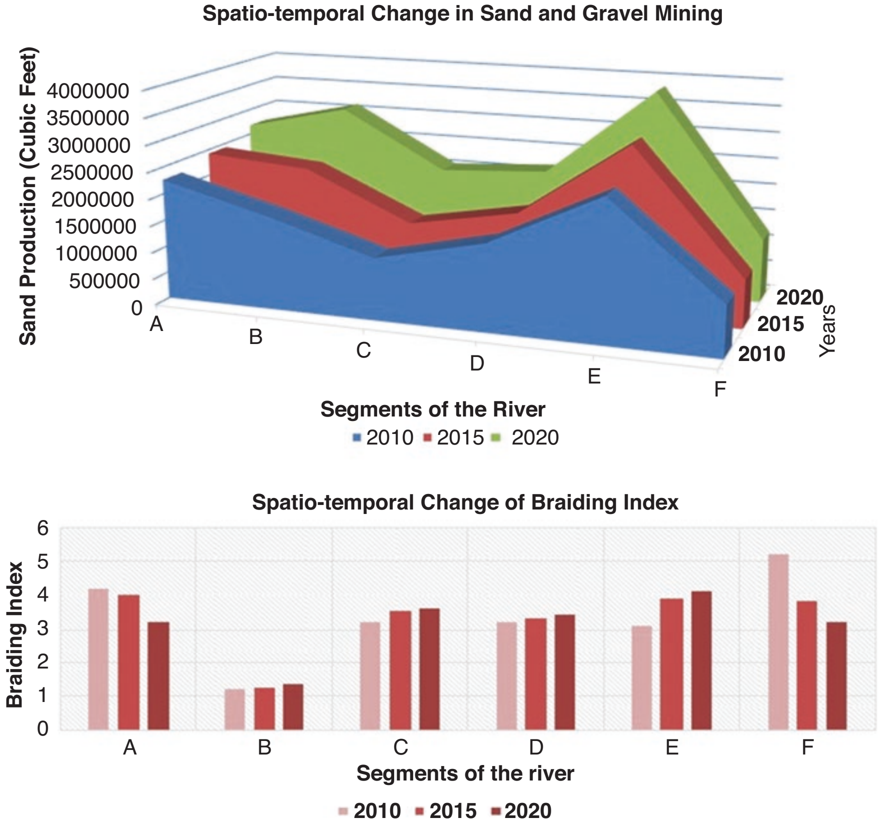

Fig. 16.5 Relationship between sand and gravel mining and braiding index

3.3 *Width Depth Ratio of the River*

The cross profiles along each mining site of the river as well as the bank position in the segments (Fig. 16.3) show that both width and depth of the river are a quite dynamic and ever-changing phenomena. The width of the river has been increased gradually in case of most of the mining sites from 2010 to 2015 and from 2015 to 2020 (Fig. 16.6) specially for uncontrolled and over mining of sand using machineries and heavy vehicles (Fig. 16.4i–j).

The long profiles of the river drawn for the years 2018–2020 reveal that aggradation is found in upper portion of Segment-A (1–14 km) and the degradation in the remaining portion of Segment-A and in the Segments B to E (14–35 km) and again aggradation in the lower most reach, that is, Segment F between 35 and 51 km (Fig. 16.7). The rapid decrease in slope at the foothill zones plays vital role in aggradation on the river bed for deposition of suspended as well as bed load of the river flow, but the degradation of the river channel in the next reaches is resulted for

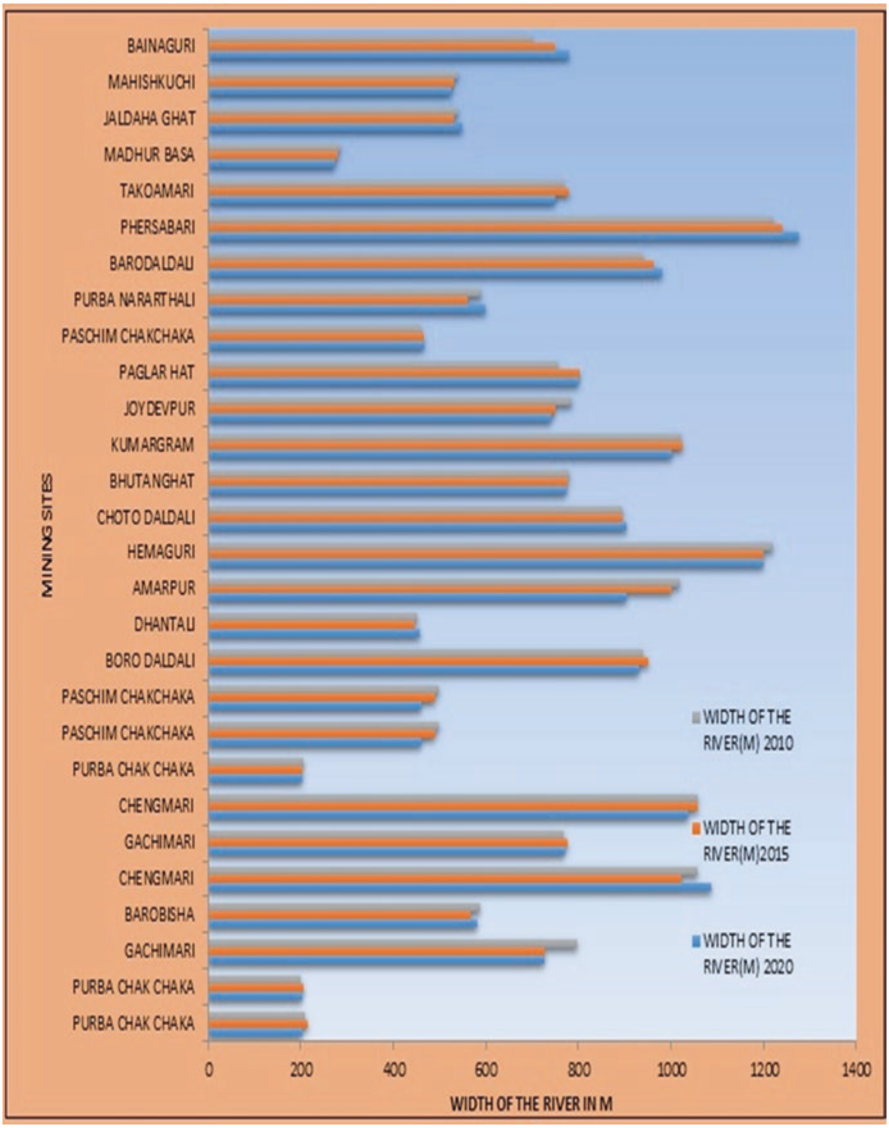

Fig. 16.6 Temporal change of width of Raidak-II river at different mining sites

irrational and over mining of the sand and gravel during this period using machineries and heavy vehicles (Fig. 16.4a–d and 16.4i–j). Finally, the aggradation recorded lower part of this river, that is, in the segment-F, for concentration of loosed materials due to mining in the upper part and subsequent deposition in the lower reach.

The investigation of mining site-wise elevation of thalweg along the river valley for the years from 2018 to 2020 shows that the elevations are decreasing except the

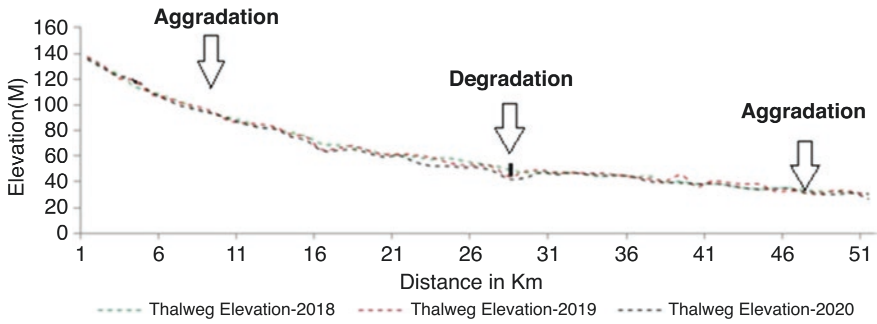

Fig. 16.7 Temporal change of long profile of Raidak-II river

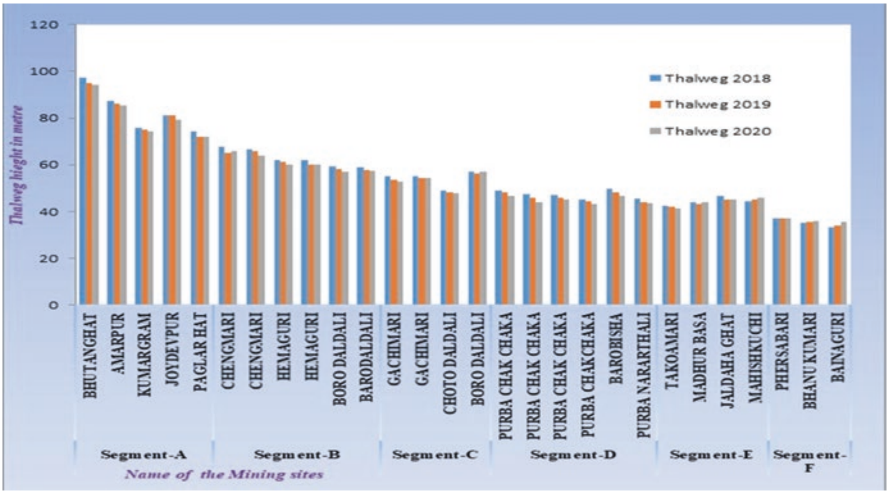

Fig. 16.8 Changing elevation of thalweg at different sand and gravel mining sites

lower part of the river, that is, segment-F where aggradation is quite evident (Fig. 16.8).

The average width depth ratio for each segment has been calculated and plotted in the Fig. 16.9 which depicts that width depth ratio is considerably decreasing from 2010 to 2015 and from 2015 to 2020 for each segment except a little higher ratio in the Segment F during 2015 than 2010 and 2015. This changing scenario of width depth ratio validates the rapid degradation of river bed due to the sand and gravel mining in the last 10 years and immediate deposition of loosed material in the lowest reach (Segment-F) (Fig. 16.10).

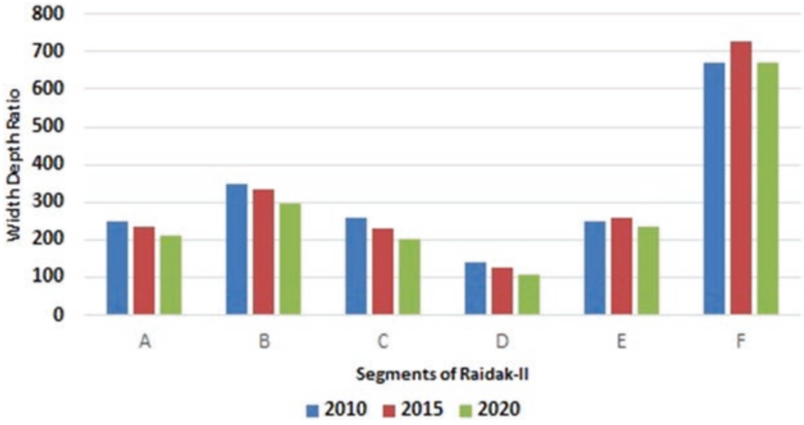

Fig. 16.9 Temporal change of segment wise width depth ratio of Raidak-II river

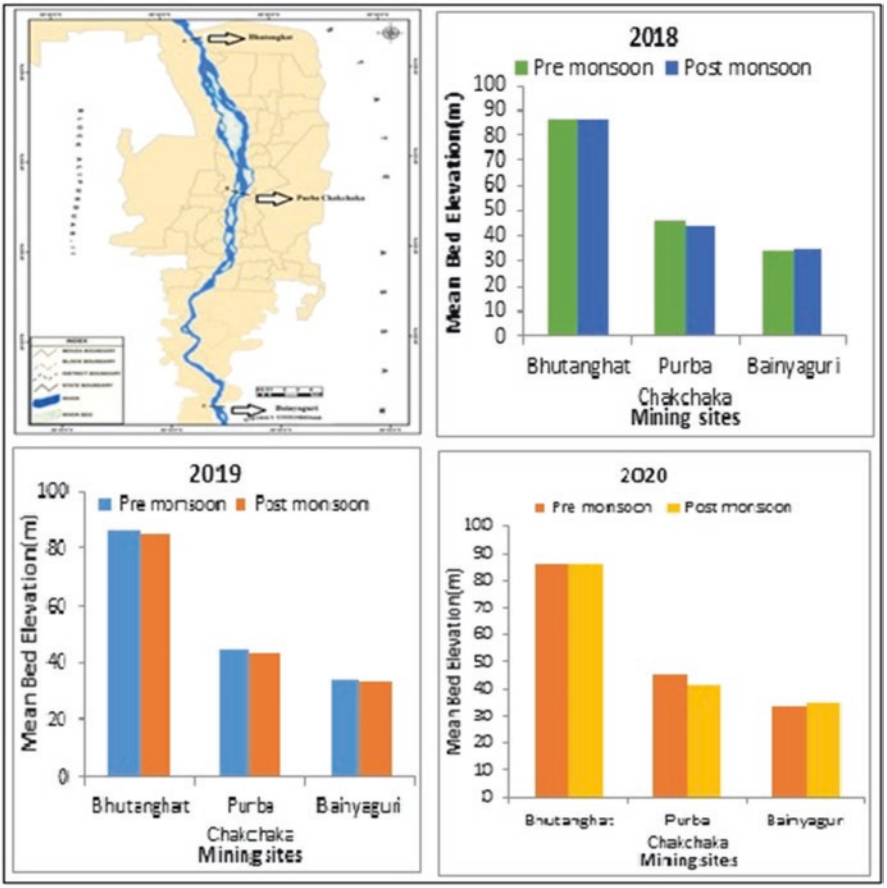

Fig. 16.10 Location of three representative sites and mean river bed elevation—2010, 2015, and 2020

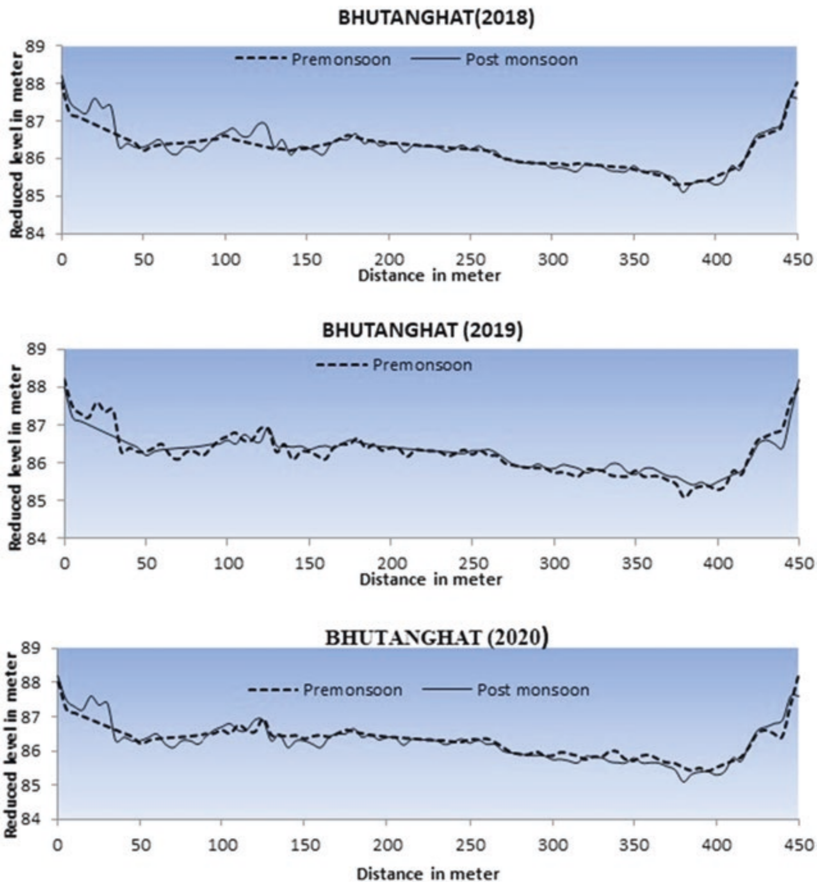

Fig. 16.11 Cross profile of Raidak-II river at Bhutanghat in 2018, 2019, and 2020

3.4 Pre-Monsoon and Post-Monsoon Variation in Elevation of the River Bed

The cross profiles of the three representative sites form the upper, middle, and lower reach at Bhutanghat, Purba Chakchaka, and Bainaguri (Figs. 16.11, 16.12, 16.13) are drawn and the mean elevations of the river bed have been calculated and finally plotted in the figure for demarcating the impact of monsoon flow in this river (Fig. 16.10). The decrease in mean elevation of the river bed has been recorded in case of upper two representative stations at Bhutan Ghat & Purba Chakchaka, whereas increase in elevation is found at Bainaguri. The sand and gravel mining from the river bed is accelerating the depth of the river in (Fig. 16.4g and h) at upper two sites compared to the rest.

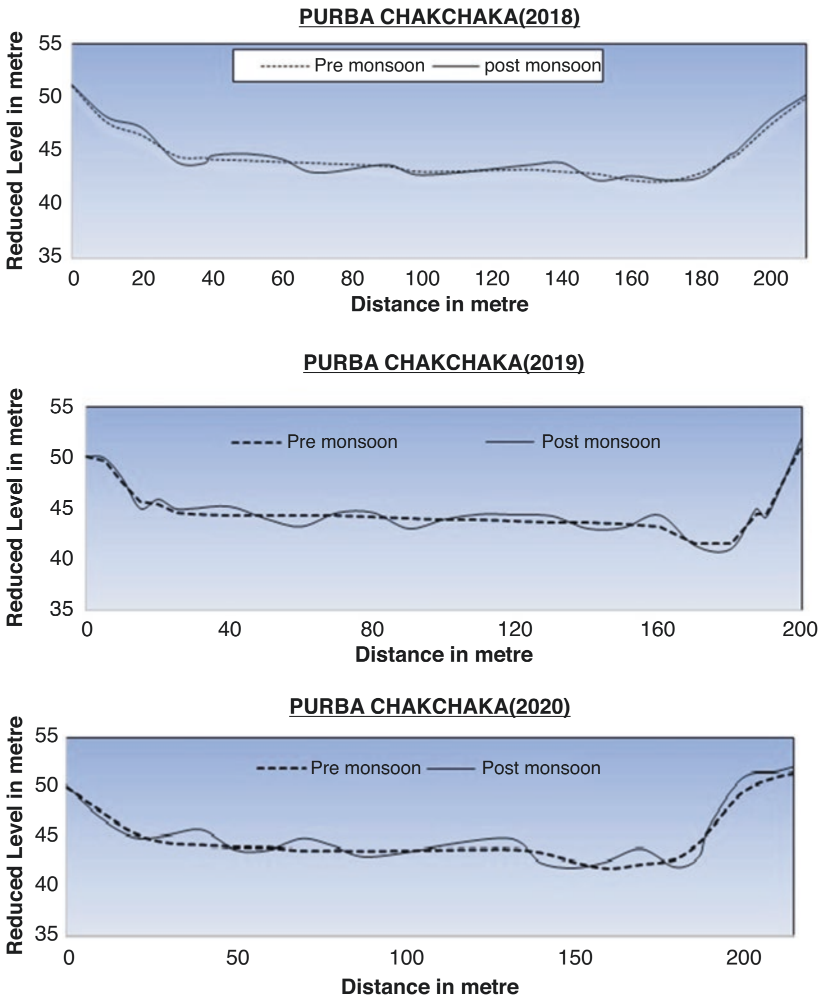

Fig. 16.12 Cross Profile of Raidak-II river at Purba Chakchaka in 2018, 2019, and 2020

3.5 Perception Assessment of the Contributing Factors

Since the initiation of human civilization, human being had been using water, sand, silt, clay, boulders, gravels, fishes, and flora and fauna, but human dominancy over the river became profound for their overuse and misuse of the natural resources available in the river. The resources of Raidak-II river are also being used by the local people residing along the riparian lands in different ways. Fishing through the year by logging bamboo and cultivation of vegetables and fruits are practiced in this river for sustaining the livelihood of the local peoples (Fig. 16.4k, l and o). Both bamboo made and concrete bridges are developed at Barobisha for communication

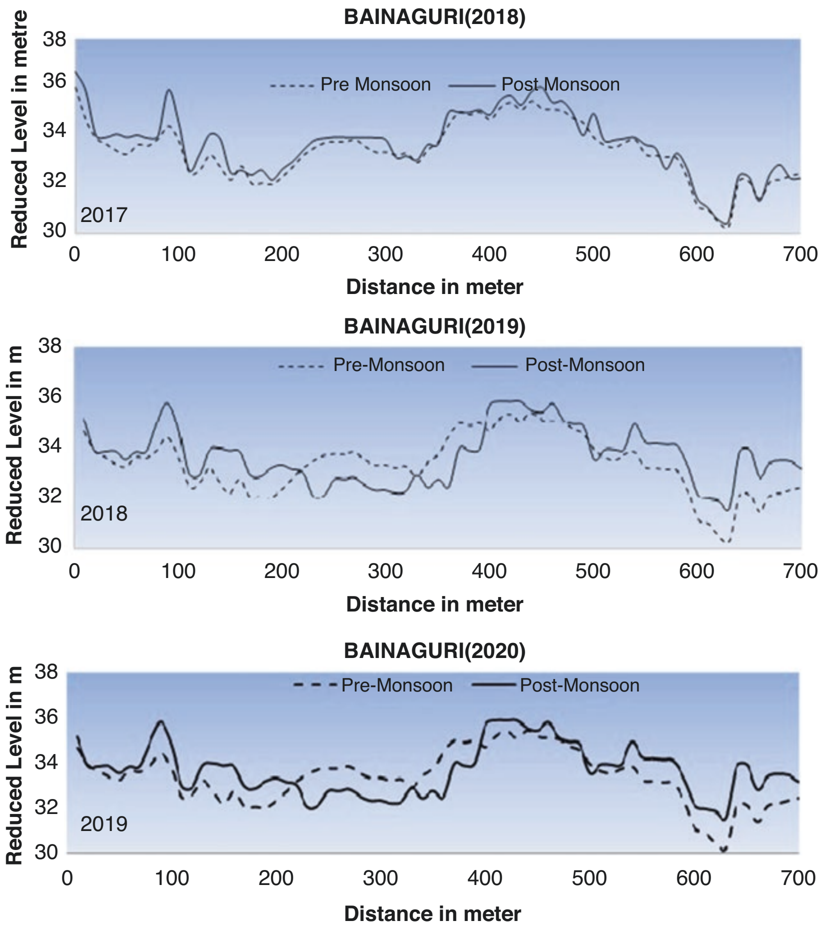

Fig. 16.13 Cross profile of Raidak-II river at Bainaguri in 2018, 2019, and 2020

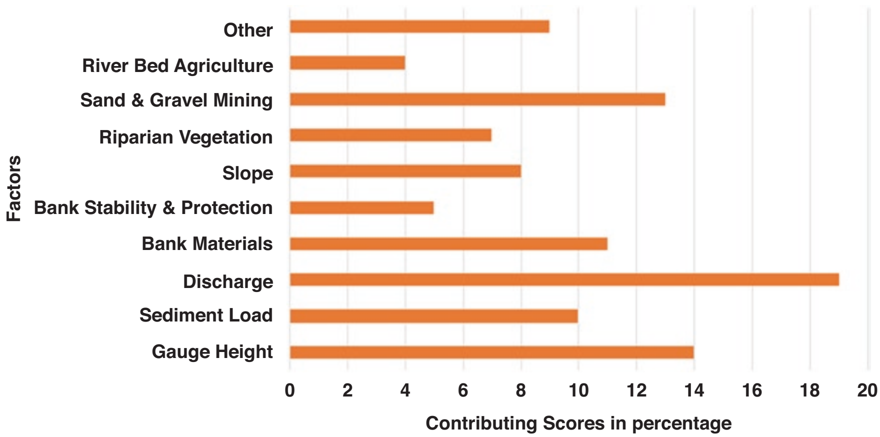

Fig. 16.14 Contributing factors of morphometric change of river Raidak-II

between both sides of the river (Fig. 16.4m). They have also tried to protect the river bank with boulder wall and vegetation (Fig. 16.4n and m). All these anthropogenic factors are affecting the morphometric character of the river with different rates. The data of different contributing factors both natural and anthropogenic factors such as gauge height, sediment load, discharge, bank materials, bank stability and protection, slope, riparian vegetation, sand and gravel mining, river bed agriculture are collected from different sources and used for analyzing rates of contributing weightage responsible for geomorphic changes of this river. The rating of these parameters is made out of 100 on the basis of expertise knowledge, field experience, and community perception. Scores of each influencing factor are plotted to visualize their contribution (Fig. 16.14). This assessment reveals that sand and gravel mining plays most profound action in changing morphometric features of this river compared to other anthropogenic activities on river.

4 Conclusion

Sand and gravel mining becomes one of the important economic activity of people of the study area as it supports livelihood of the people residing in nearby river banks and involving in this activity as labor, drivers and boatman, contractor and businessman. Thus, it causes a tremendous craze for earning based on the river. Moreover, the leaseholders of the mining sites are involved in irrational mining beyond their limitation as mentioned in the lease deed. Some people are involved in occupying river beds for unauthorized mining using their local and political influence. Considering the pros and cons of sand and gravel mining from the socioeconomic point of view, it can only will be sustainable activity if it can be practiced in a balanced way considering the offer of nature or river in their concave bank or only in the higher bars which will be replenished in the next season. The present work thus provokes to pursue more research work on sand and gravel mining related issues to make this economic activity more sustainable in the future.

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Chapter 17
Resuscitating the Regional Rivers: A Crusade against Cauvery Crisis and Coloured Legislations

Sasi Varadharajan

Abstract In this global circuit of economies and ecologies, a balance of nature and future is a necessary mandate. However, the true spirit of conservation has been lost somewhere down the hypocritical lane of cultural affliction and commercialization. One such resource that is ravished inside the regime of conservatory laws is the river sand. In-river mining for this precious minor mineral has surpassed every limit—legal, social, economical, hydrological and geomorphological. Asthmatic locals, unstable bank sides, famished farm lands and the excavation pits that look like wells speak volumes about the alarming threat to riverine ecology. The existing constitutional laws and conservative measures too are coloured without sustainable spirit; failure to account for the hydro-geomorphology, sand inflow, basal flow and irrigation dependency in the respective State rules has reduced our regional rivers into rampaged deserts. The paper, here, outlines the field-level observations in the Southern river Cauvery to map the grassroot level implementation gaps of EIA Notifications and TNMMC Rules (Tamilnadu Minor Mineral Concession Rules); advocates for advanced legal-scientific convergence to fix the allowed mining depths, spatial extent and mechanical machineries and finally suggests for a renewed umbrella framework legislation on the lines of the 2016 Sustainable Mining Guidelines to surmount this man-made disaster.

Keywords Sand quarry · In-river mining · Dredging limit · Ecology · Impact assessment

1 Introduction

In-river mining 'of' sand is always flaunted as a defensive conservation measure against excessive siltation in river beds; however, the mining 'for' sand captures the mineral and nutritious essence of the riverine ecosystem, rupturing the already

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delicate food chain. Regional rivers in several states throughout the Indian land have changed their courses, depriving the dependent agricultural farms of their usual water source; they dry up more quickly in the summers because of decreasing base flow. Recently, the legal studies combined with scientific analyses of river bed, triggered by the seminal literature of Mathias Kondolf on hydro-geomorphology among others, have brought the age-old environmental evil into limelight.[1] The rampant sand mining has interfered with and, in turn, invariably altered the river drainage patterns, channel morphology, irrigable areas, etc. There have been several crusades to save the regional rivers against degrading dredging that have been launched by the local populations especially medium- and small-scale farmers who are affected the worst, an example being several PILs[2] filed against mining in the Southern river Cauvery.[3] The legal rule books on this issue are State varied since sand, as a minor mineral, falls under the List II of seventh Schedule of the Indian Constitution.[4] The focus here is to study the environmental, social, and economic effects of in-river sand mining by the method of field study done in the particular sand quarries of Mohanur and Oruvanthur situated on the river Cauvery in the Namakkal district of the State of Tamilnadu[5]; to compare the legislative innovations and lacunae in the State-wise Mining Rules to stress the need for Central intervention; to analyse the possibilities of alternative construction materials; to finally arrive at socially acceptable green suggestions as sustainable measures against this corroding menace.

2 River Sand: Economical and Ecological Importance

Natural sand from water channels, especially rivers, is the most desirable construction material for their easy extraction and readily usable properties among others. Unlike artificial sand, natural sand is already washed, exposed, naturally graded and treated under external environmental pressures making it more durable and

[1] It is pertinent to note that most of the sand mining papers are inspired by the 'hungry water effect' phenomenon propounded by Kondolf to study the consequential erosion caused by sediment-starved water in rivers; though the original phenomenon was around dredging in reservoir area, the same has been arguably extended to dredging inside rivers along and across their course. Other notable literature supporting innovative legal restrictions on volume of sand mined are flood control and sand properties study of Kondolf, the recent emerging ideas of mapping ex-sand mining areas through high resolution satellite imageries, data on successful experimental application of sand alternatives in construction, etc. (Kondolf 1997, Elavenil et al. 2017, Novie Indriasari 2018).

[2] 'Public Interest Litigations'.

[3] Varadharajan v The District Collector and Ors [WP. No. 22433 of 2017].

[4] Entries 23 and 64 of List II of seventh Schedule of Indian Constitution coupled with sec.15 of the Mines and Minerals (Development and Regulation) Act.

[5] These quarries on the river Cauvery are not currently under operation due to the efforts of the local villagers and judicial intervention through the PILs filed; the data used herein are collected by the field observation done over the period of 2 years (2016 - 2017), further corroborated by the historic images obtained from Google Earth technology.

requiring lesser refining than manufactured sand (Elavenil et al. 2017); it is preferable to sea sand because of the low chloride content and hence, will not rust the iron pipes used abundantly in construction. The cost of extracting sand from inland water bodies is cheaper in several states than artificially manufacturing because of the low royalties and liberal environmental sanctions sadly prevalent.

In terms of ecology, sand plays several significant roles beginning with ground water recharge. These fine aggregates retain water in them even during dry seasons, playing a vital role in basal flow maintenance, being nature's own aquifer. During excess run-off in monsoon, the sediment deposits along with bank-side vegetation build resistance against the water speed, dissipating energy, thereby preventing erosion and flooding (Padmalal and Maya 2014). Also, the nutrient elements in the river sand and the detritus thereof are the natural food of several infauna and epifauna populations (Sheeba 2009); the natural geochemical composition of the fluvial, alluvial and colluvial deposits along the flood plains make them more suitable for agriculture, requiring very less chemical supplements.

3 Epidemic Effects of Mindless Mining

As stated above, river sand is economically significant as it is used as a common construction material in mortar; however, it is environmentally expensive for maintaining the ecological balance in riverine ecosystems and promoting agriculture in fertile floodplains; this calls for a cautious use of this natural resource. But, mindless of the green effects of mining, quarries excavate sand below permissible limits (depth wise and extent wise) in most of the regional rivers in India. Though the legalities of the mining leases vary from state to state, the ill effects of the same are common as follows.

During the monsoon seasons, water flow increases and so does the river speed; in mined out water beds, the resulting sediment starvation adds to the river velocity which causes the water channels to erode the banks, standing superstructures, the already mined river bed as well washing away whatever soil aggregates remain. This hungry water effect coupled with deep pits from mining pave way for head cutting and the knick points thereof, which further migrate upstream with every heavy flow (Kondolf 1997). Farther the knick points move upstream, larger the variation in bed slope is; changes in river bed slope affect the drainage pattern of the river, for example, increased downward slope makes the river to drain off faster, leaving little time to catch and conserve the run-off for agricultural and other uses; this also slows down the rate of groundwater recharge as more stable running water can percolate down the ground than the fast run-off. Since the basal flow is dependent on factors like soil type, surface soil moisture, bed slope among others (Jia Qin et al. 2017), removal of fine aggregates and altering the slope of river bed result in a drop in base flow, worsening the water scarcity in summer season.

The hungry water also erodes the water structures like pumping stations and water tanks of the public works department that have to be situated along the river banks to satisfy public water needs and other transportational structures like bridges, across-the-river rail tracks, thereby jeopardizing public safety, connectivity, etc. Erosion also destabilises and undercuts the bank sides (Kondolf 1997) leading to bank collapse,[6] bank slides and bank shifts along with loss of riparian buffer zones which may force the dependent species to migrate further into human habitations, especially when the protected or reserve forest cover runs alongside the river course. For example, The Nipah Virus outbreak in Kerala can be attributed to migration of the fruit bat species from its original habitat to human settlements because of diminishing riverside/backwater vegetation cover.[7] In the rivers running parallel to the sea coast, bank destabilization can also account for salt water intrusion making the waters unfit for human consumption.

Extreme effects of in-river sand mining are the changes in the hydrology and geomorphology of the river, its physical and chemical properties like the turbidity levels, course of the river, its flow velocity and bed sediment morphometric characteristics (Sadeghi et al. 2008), etc. Increased mining activities on one side of the river may leave the other bank at a higher elevation and the water flow will be more only in the low-lying bank; this change in river course severely starves the nearby agricultural lands as most of them are dependent on river water irrigation than their own wells or other means[8]; it pulls down further the already low agricultural productivity. This will be worsened if only the main river is mined and not its distributing channels; because it may cause the river and the channels to stand at different elevations—with the distributaries at a higher elevation than the main river; as per the basic laws of gravity and physics, water cannot flow naturally from the low-lying river into the high-standing channels unless they are dredged to equal heights.

Also, the mining is not uniform over the leased river area resulting in irregular deep pits of varying depths; in dry seasons, when the flow volume is already low, water has to first fill these pits before resuming its flow further. This results in the water not reaching the far-off draining points, thereby again depriving the associated agricultural irrigation systems of their major source. Apart from hampering the on-time irrigation of farm lands, the pits also stagnate water acting as a breeding ground for mosquitoes and other insects (the stagnation illustrated clearly in the Figures below). Lack of uniformity in the extent and depth of dredging may lead to widening and deepening of the river beds; increase in the width of the river expedites the vaporisation and decreases ground water recharge when there is little flow in the

[6] Banks along the river mouths close to sea are more vulnerable to collapse.

[7] Vinod A. Narayan, 'Nipah Virus Outbreak in India: Is it a Bat-Man Conflict?' (2019), *International Journal of Community Medicine and Public Health* 6(4), 1826-1830;

Damayanti Datta, 'Uncontrolled Sand Mining Led to Kerala Floods', *dailyO*, Aug 27, 2018, https://www.dailyo.in/variety/kerala-floods-sand-mining-sand-mafia-illegal-mining-soil-erosion-rivers/story/1/26297.html

[8] In fact, most farmers are marginal and small scale that they cannot afford to dig wells of their own to irrigate lands.

non-monsoonal periods. Uneven mining and increased erosion may also lead to sinkhole formation, especially in salt bed areas and limestone landscapes.[9]

Increased turbidity levels in water from the sand excavation have toxic effects on the aquatic fauna like blocking of gills in fish, lowered light penetration and dissolved oxygen levels, poor spawning, etc. Since the organic detritus also gets mined along with sand, the natural feed of the infauna and epifauna populations like molluscs is cut off. In turn, the animals that feed on molluscs like crabs, fish and birds suffer. Also, the larval forms and nymphs of several terrestrial organisms like dragon flies, frogs are aquatic; since they feed on mosquitoes and other harmful insects, their depletion will leave the risk of infectious diseases unchecked (Sheeba 2009; Kondolf et al. 2002). Cutting off the feed of micro fauna which are at the bottom of food pyramid disrupts all the levels in turn, up to the top of the food chain. The stagnation in pits and low water flow also advances the growth of aquatic weed like water hyacinth and algae cutting off sunlight and oxygen for fish. Thus, invasive in-river mining for sand is detrimental to the delicate ecological balance in the riparian buffers and also destroys the conventional food web. It is not just the riverine environment, even the ecology of hinterlands is affected by endless quarrying. The trucks carrying sand are, often, not covered properly and when the moisture in the scooped-up sand dries off with the heat during transportation, the finer particles fly off increasing the level of suspended particulate material in air (Naveen 2012); this lowers the vision in roads and also makes it difficult for breathing, with the two-wheelers suffering the worst. Constant transportation in densely populated areas using common roads accessed by citizens dangers the risk of road accidents and also leaves the rarely laid roads in rural areas totally potholed. The fly offs from the trucks are thickly deposited in the plants and residences of those located closely to the checkpoint or the quarries, leaving them highly vulnerable to respiratory illnesses.

Inside the quarries, bunds are constructed to transport the material mined from the river to the outside; these bunds, used for connectivity, when constructed in large number and across the river, completely cut off free flow of water, forcing the river to change its course and flow through the places where there are no bunds.[10] When there is sudden increase in the water volume due to rains or released excess from reservoirs, the velocity of the river picks up and causes channel incision on the sides due to the obstructing bunds (Kondolf 1997). When the river is already dredged and the slope has become downward, the rushing flow of sediment-starved water will increase the basal sheer stress increasing the chances of incisions. Also, when the bunds restrict the flow of water on the sides where there are distributing canals, this

[9] 'Sinkholes', *Water Science School (USGS)*, May 5, 2020, 18:50 PM, https://www.usgs.gov/special-topic/water-science-school/science/sinkholes?qt-science_center_objects=0#qt-science_center_objects

https://www.conserve-energy-future.com/causes-effects-and-types-of-sinkholes.php

[10] This phenomenon where the river has to change its course because of obstruction falls under the concept of 'river channel relocation' (Alissa Flatley et al. 2018).

will cut off water for irrigation and actively create inactive water channels which were totally dependent on the river water inflow.

Thus, changes and interferences imposed by dredging put the ecologies and the human population in jeopardy by altering agricultural drainage patterns, affecting the micro and macro fauna populations and destroying the riverside vegetation.

4 Field Observation in River Cauvery

Cauvery, a perennial river of South India, originates in the Western Ghats of Karnataka, irrigates the farm lands of Tamilnadu and drains into Bay of Bengal. Being the largest river in the State of Tamilnadu, it is a major source of water consumption, hydro-power generation and irrigation in several districts[11]; unfortunately, the river is also a target of sand mining due to its rich alluvium. The field research and observation in two sand quarries across the Cauvery river—Mohanur and Oruvanthur of Namakkal district, Tamilnadu have corroborated the already stated undesirable and catastrophic effects of in-river mining while revealing a few other disturbances.

In Fig. 17.1, the name board with details to be provided under the TNMMC Rules[12] and the Environmental Clearance (EC) given for the quarry are visible. But, the contact of the Asst. Superintending Engineer and other officials for the sand mine, as stated in the Board, was not functional leaving the public (especially local farmers) no means to complain the noted transgressions and ask for necessary details; this highly undermines the spirit of community participation. And the improper maintenance had led to the invasion of *Prosopis juliflora* species which tends to suck the already low ground water level.

As shown in Figs. 17.2 and 17.3, these connecting roads block the free flow of water and are in clear violation of the conditional EC granted by the State Environmental Impact Assessment Authority (SEIAA); there were several roads resembling this during the field visit, giving a wing road system for the trucks to move easily to and from the river[13]; there were no enough pipes across these bunds and roads to allow water from one side to flow through the other as well. This, coupled with excess mining on one side of the bank, has forced the river to change its course in this area (further explained in the Google Earth Images). Also, since the two quarries studied here (Mohanur and Oruvanthur) were close to each other, even

[11] 'Basin Details: Cauvery and Southern Rivers Organisation', Central Water Commission, May 1, 2020, 14:40 PM
http://cwc.gov.in/csro/about-basins

[12] Tamilnadu Minor Mineral Concession Rules, 1959.

[13] The number of bunds and connecting roads were out of the necessary proportion to allow the bulk no. of lorries to come and go without having to take reverse or face any discomfort; clearly, the proportionate numbers necessary to transport were not maintained, rather, convenient numbers took prominence while laying bunds and deciding their width and lengths.

Fig. 17.1 Photograph of the quarry name board erected by the Public Works Department in Mohanur, Nammakal district, Tamilnadu. Source Credit: Author (taken during the field visits 2016–2017)

Fig. 17.2 Photograph of a connecting road laid inside the river Cauvery for transporting the sand from dredger to the bank side, Mohanur. Source Credit: Author (taken during the field visits 2016–2017)

Fig. 17.3 Photograph showing the various bunds constructed to transport sand while restricting water flow, Mohanur. Source Credit: Author (taken during the field visits 2016–2017)

after the mining lease expired for the Mohanur quarry, the bunds and connecting roads therein were not destroyed, rather, they were used to travel to the Oruvanthur quarry. As a result, a separate transport mode was artificially arranged inside the river where the trucks need not use the main village roads or lay a new access road from the village road to Oruvanthur quarry; the trucks used the access road of Mohanur quarry from the village and then travelled inside the river itself using the bunds and inroads laid down when the Mohanur quarry was under operation, to reach the Oruvanthur quarry. This was largely because of the reason that they could not lay a new access road from the village to the Oruvanthur quarry as the proposed route cut through protected forest area. So, instead, they chose to have a road inside the river blocking the water flow and its usual course.

From Figs. 17.4 to 17.6, it can be seen that the depth of mined area is more than 1 m (3 feet) which is the criteria under TNMMC Rules[14]; the actual depth varied from 10 to 15 feet depending on the desirability of the sand in that particular area. Also, the uneven mining has converted the entire Cauvery river bed in this region into a zone of pits, rather small wells all over the place. This stagnates water and also prevents the free water flow as the incoming water has to first fill these deep pits

[14] However, it is contended that the 1 metre rule does not start from the surface but from the bed level after removing shoals above. Even that way, several pits in Mohanur quarry have surpassed that limit. As per the recommendations in *Deepak kumar & Ors v State of Haryana & Ors* [(2012) 4 SCC 629] and 2013 EIA Notification, the limit laid down is 3 metre/water level whichever comes earlier.

Fig. 17.4 Photograph showing a sample of the pits created due to uneven and in-depth mining, Mohanur. Source Credit: Author (taken during the field visits 2016–2017)

Fig. 17.5 Photograph showing the difference in elevations inside the river due to uneven mining, Mohanur. Source Credit: Author (taken during the field visits 2016–2017)

Fig. 17.6 Photograph portraying the depth of pits post-mining, which are more than 10 feet, Mohanur. Source Credit: Author (taken during the field visits 2016–2017.

Fig. 17.7 A photograph showing mechanical dredging using bull dozers as against manual mining under the TNMMC Rules and other Sand Mining Guidelines, Mohanur. Source Credit: Author (taken during the field visits 2016–2017)

Fig. 17.8 A photograph showing the invasive aquatic weeds like water hyacinth closing the water surface, Mohanur. Source Credit: Author (taken during the field visits 2016–2017)

before resuming its flow; this shortens the distance reachable by the water during dry seasons. From Fig. 17.5, it is evident that mining only in a few parts of the river leaves the other areas to stand at a higher elevation, as a result, the river bed is highly uneven (sometimes with height difference of 3 feet or above) forcing the river to flow in the low-lying portions, leaving a vast portion of the river dry; if the mining happens on one bank of the river alone, the water will tend to flow towards this low-lying bank because of the deep dredging and the opposite bank and associated farm lands will have to suffer.

As per the TNMMC Rules applicable to dredging in the state, only manual mining is permitted; however, through a judgment by J Banumathi of the Madurai Bench of the Madras High Court, a maximum of two mechanical dredgers like poclains can be used if the terrain is rugged and after getting due permission from the District Collector. This has been misused by the lease holders to always use poclains regardless of the surface texture and the number is always more than 2. Using these dredgers is one of the reasons why the 1 m limit is not maintained. Manual mining will give seasonal job opportunities to the local public while ensuring the depth restrictions. Also, the deep pits created by poclains are not covered at the cessation of operations leaving stagnant water and encouraging the growth of water weeds, depriving the fish of oxygen and sunlight as shown in Fig. 17.8.

From Figs. 17.9, 17.10, 17.11, 17.12, clear violation of the minimum safe distance of 500 m[15] from the quarry sites and standing superstructures like bridges, water tanks, etc., can be seen. This is because of the invasive mining against the

[15] As per the 2020 Enforcement and Monitoring Guidelines for Sand Mining, a safe distance of 1 km from major bridges/highways, etc., to be maintained where no sand or gravel mining shall take place.

Fig. 17.9 A photograph showing the close proximity of dredging operations to the electric power grids, Mohanur. Source Credit: Author (taken during the field visits 2016–2017)

Fig. 17.10 A photograph showing the erosion of standing superstructure close to the sand quarry, Mohanur. Source Credit: Author (taken during the field visits 2016–2017)

Fig. 17.11 The photograph shows the erosion and de-stabilisation of a public water tank in the river Cauvery near the quarry site, Mohanur. Source Credit: Author (taken during the field visits 2016–2017)

Fig. 17.12 Photograph with a close view of the effect of sand quarrying near superstructures across the river, Mohanur. Source Credit: Author (taken during the field visits 2016–2017)

Fig. 17.13 Photograph showing a dry distributing canal with the sluice gates, Oruvanthur. Source Credit: Author (taken during the field visits 2016–2017)

proposed plan submitted for the EC to SEIAA.[16] These on-site impacts also evidence the lack of field monitoring by the hydrology, geology and public works department, etc. More often than not, sub-leaseholders go beyond the limits of the lease area to dredge desirable fine aggregates from nearby sand carpets, without much regard for these essential superstructures. Close to the Mohanur quarry, there is an across-the-river bridge, railway track and public water tanks and pumping stations. As seen in Fig. 17.13 uneven mining makes the river to lie at a lower elevation than the sluices of distributing canals thereby cutting off water inflow into them; it is to be noted here that this sluice canal present near Oruvanthur quarry site is crucial for irrigating several farm lands in both Namakkal and Trichy districts.

Not only these impacts, the sand quarried on these sites was sold at exorbitant prices with no share in profits for the government. Technically, the Government was subleasing the quarry sites through its PWD department instead of doing it on their own; the leaseholders paid only royalty and fixed fee for every truckload of sand with the EC being obtained by the PWD itself; usually, the number of trucks is understated and the dug-out sand is held in private sale points, trafficked to other states where there is moratorium on in-river mining of sand; by the time it reaches the common men, the truckload will be worth 7 to 10 times higher.[17] In the recent

[16] 'State Level Environmental Impact Assessment Authority'.

[17] It has been stated by the local farmers that a truckload with 2 units of sand was fixed around 600 to 650 rupees by the Government in Mohanur sand quarry but was sold around 4500 rupees to oth-

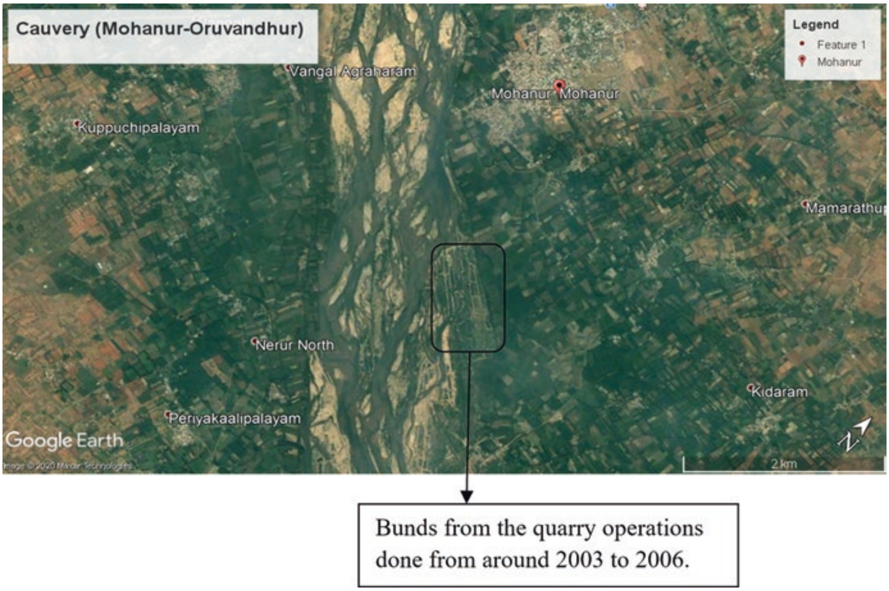

Fig. 17.14 Google Earth Imagery from 2006 shows the water flow, direction of the river Cauvery near the Mohanur village and the quarry site where the bunds from previous quarry operations are still visible and not removed

years, the State Government has announced that the quarrying operations will be done by.

the Govt itself without subleasing and people can book units of sand needed through the Govt portal ‘TN Sand’. However, even this is sceptical, as PWD being an arm of the Government, submits its prefeasibility reports and others to get approval from another arm of the Government, SEIAA. And the lack of public consultation for quarries falling under B2 projects severely undermines the transparency and integrity of the process.

The change in river course over the years near these quarry sites in Cauvery can be discerned from the historical images taken from Google Earth Imagery.

Comparing Figs. 17.14 to 17.20, it can be seen that the river has shifted to its left side near the quarry site; this can be attributed to the unremoved bunds that restricted and altered the flow over the years and the in-depth mining on the left side leaving pits of varying heights (till 10 feet).

Whereas the Google Earth Imageries near Oruvanthur from 2006 to 2020 (as shown in Figs. 17.21 to 17.26) show the migration of river towards its right side.

The Google Imageries from 2006 to 2019 in both Mohanur and Oruvanthur sites, on close observation, also indicate that even when the dredging was ongoing only in Mohanur quarry during the periods of 2014 to 2016, the leaseholders had gone

ers by the sub-lease holders. This was of course the minimum price where the private profiteers sell them at 5 to 6 times this price.

Fig. 17.15 Google Earth Imagery from 2010 shows that the river course has remained the same yet the bunds have not yet been removed despite there being no quarry operations for almost 5 years

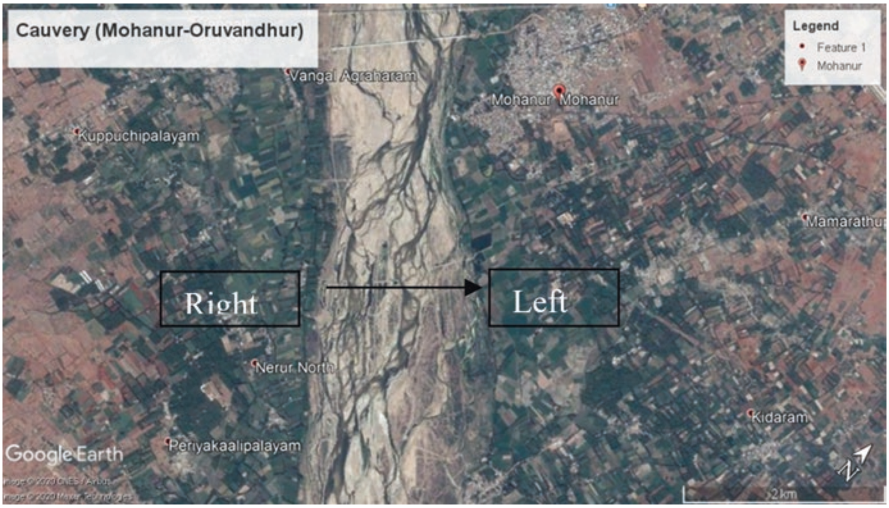

Fig. 17.16 Google Earth Imagery from the summer of 2014 shows the drainage pattern of river in the area, wherein the flow is almost equal on both banks

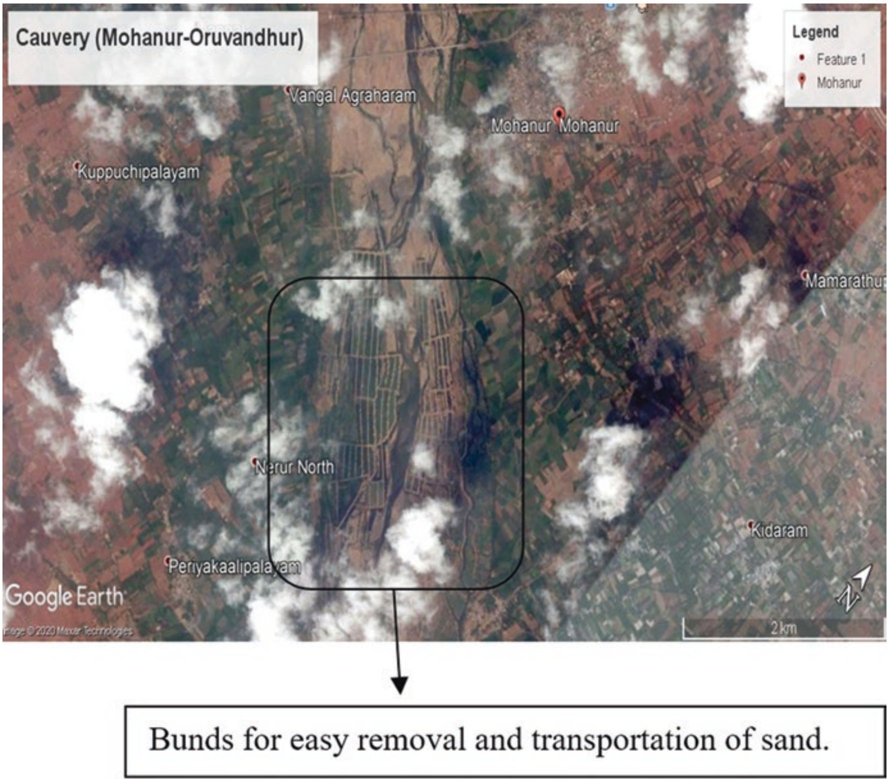

Fig. 17.17 Google Earth Imagery from 2015 where the Mohanur quarry was under operation shows the numerous bunds on both sides of the river severely constricting its flow to the middle, depriving the farms, canals , water tanks and pumping stations on the sides of water source

Fig. 17.18 Google Earth Imagery from 2016 shows that increased water flow has broken down some bunds in the quarry site; yet run-off has not reached the sides

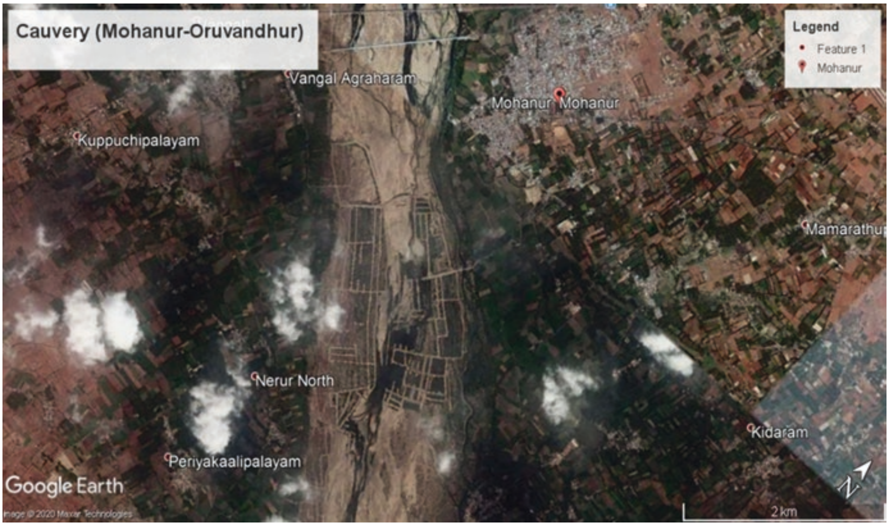

Fig. 17.19 Google Earth Imagery from 2017 shows that despite the cessation of quarry operations, the bunds have not been demolished in the site, Mohanur

Fig. 17.20 Google Earth Imagery from summer 2019 vividly shows that the bunds have not yet been removed and the river has shifted towards its left in the middle

Fig. 17.21 Google Earth imagery from 2006 shows that the river flows near both the banks, irrigating also the Oruvanthur side where a small protected forest area is located

Fig. 17.22 Google Earth Imagery from 2011 where the status quo is almost the same as 2006 near Oruvanthur

Fig. 17.23 Google Earth Imagery from 2015 shows that branch of river irrigating Oruvanthur side has disappeared; from the location of bunds in Fig. 17.22 and this, it can be seen mining in those areas have deepened the river in the middle and right, making the left side a little higher, thereby cutting off flow to that side

Fig. 17.24 Google Earth Imagery from 2016

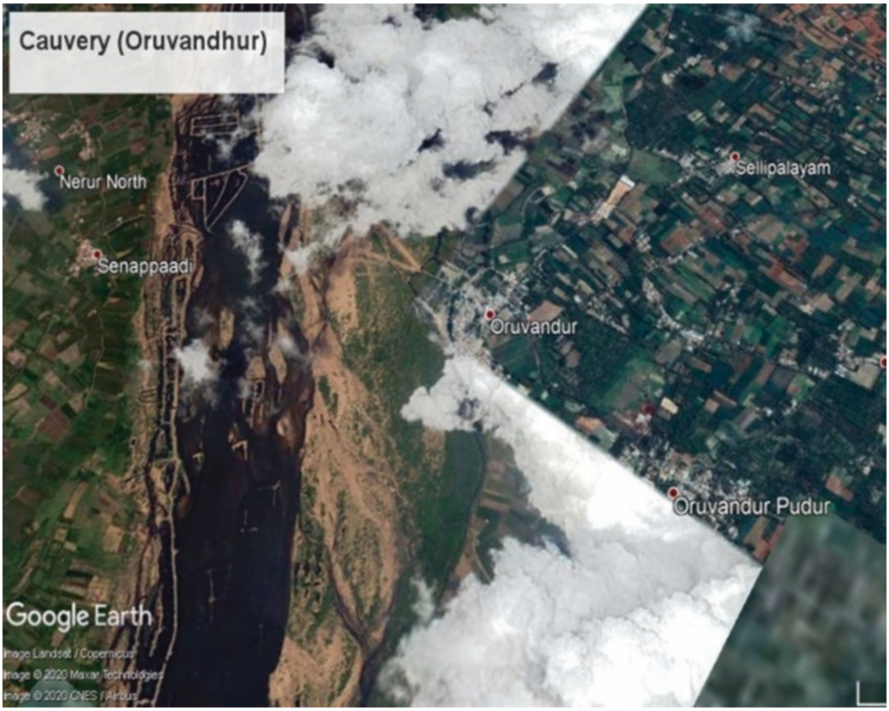

Fig. 17.25 Google Earth Imagery from 2017

beyond the lease area and have dredged portions of Oruvanthur (on either banks, especially in the middle and right side of the river) as well. This has contributed to the shifting of the river to its right near the Oruvanthur village area, even before the proposed quarry in Oruvanthur (around 2016) took place.[18]

[18] For the reasons that the river bed is uneven with changed river course and that Mohanur quarry leaseholders had already dredged beyond their permitted limits intruding into Oruvanthur region, the local farmers and villagers had heavily objected to the dredging of Raayangar Thittu, the region's left over sand carpet in 2016 to 2017; the heavy opposition coupled with Public Interest Litigation (PIL) and grass root level awareness meetings have forced the district administration to abandon the quarry operations for now in Oruvanthur.

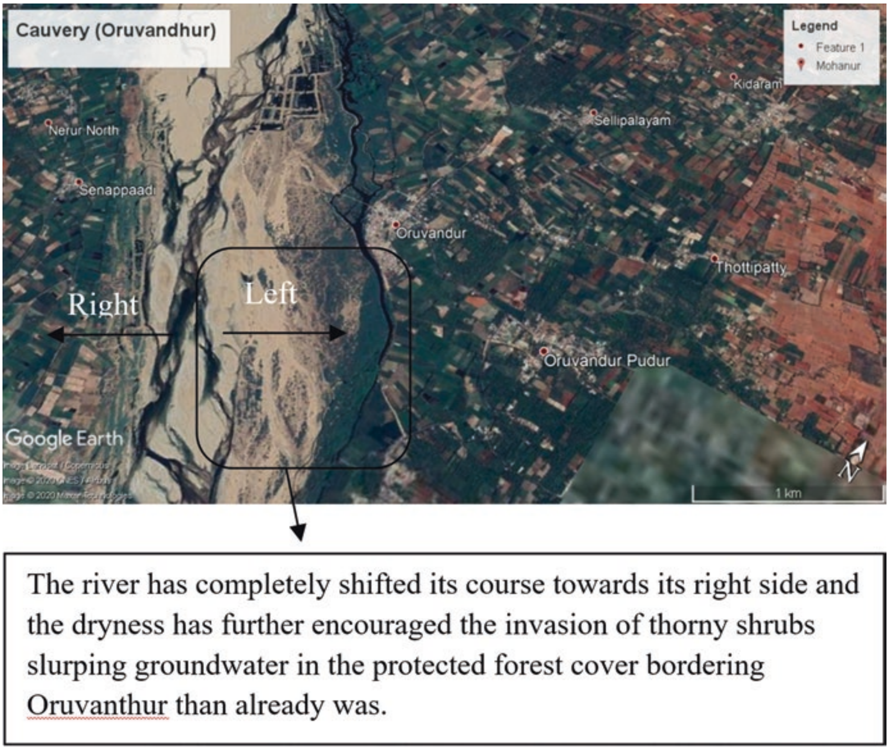

Fig. 17.26 Google Earth Imagery from Summer 2019; comparison of Fig. 17.21 and this shows that the left-side branch of the river Cauvery irrigating the Oruvanthur area has completely disappeared, starving the associated farm lands of direct water source

5 Regional Efforts to Revive the Ecologies

Sand is a minor mineral as per the MMDR Act[19,20] and falls within the purview of State Legislatures as per the List II of seventh Schedule of the Indian Constitution. With the Quasi-federalist concept in mind, the Central Government has often

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[19] It is to pertinent to note that the classification of major and minor minerals under the Mines and Minerals (Development and Regulation) Act seems to be based on their monetary value, without accounting for their depleting availability or lowered renewable rate or their significant ecological value. Even though the Central Government has the power to announce any mineral as 'major' via a Gazette Notification, there are no guidelines or criteria for doing so. A provision as to conversion of minor mineral like 'ordinary sand' into major mineral on the event of them being ravished and pushed to an all-time low renewability and the consequential affliction of water crisis or ecology damage will guarantee sound management of the Earth's limited resources as major minerals have scope for stricter supervision under one common control.

[20] As per Rule 70 of the Mineral Concession Rule, 1960 - 'Ordinary sand', as included in the list of minor minerals, refer to naturally occurring sand except the sand for following purposes viz., purpose of refractory and manufacture of ceramic, metallurgical purposes, optical purposes, stowing in coal mines, manufacture of silvicrete cement, sodium silicate, pottery and glass.

attempted to regulate sand mining uniformly in the States by providing revised guidelines without encroaching the respective legislative authority of the States. However, the regional conservation efforts lie largely in the hands of concerned State departments. Some examples of the sand mining practiced in different states are:

By virtue of Sec. 23C of the MMDR Act, the State governments have the right to fix royalties per load of mined sand and these differ from state to state, promoting trafficking of sand from states with low royalties and restrictions to where those are higher. The authorities involved in monitoring the mining and conferring licenses also vary, for example, the PWD in Tamilnadu,[21] Revenue and forest department in Assam, Directorate of Geology in Karnataka, Andhra and Madhya Pradesh, etc. The lease periods also differ from short mining contracts of up to 2 years to long-term mining leases of up to 15 years in Assam. The spatial extent of the leased area is regulated with ceiling limits in some states where a few others like Madhya Pradesh and Assam have no upper limits on the area leased; states like Haryana also impose no restriction on the maximum holding of mining areas by a single entity within the state. Not just the undesirable practices, even desirable legal provisions are not uniform; only a few states mandate the Gram Sabha, Maritime Board, Ground Water Development agency and other essential bodies' approval for sand mining.

Not just this, a study of one of the District Survey Reports (DSR) prepared by the team from IIT Madras with the assistance of State Geology Department and others for the Namakkal district in Tamilnadu exposed several inconsistencies[22]; as part of this Mineral Study, details of sand mined, money earned has been tabulated, but the sources of such data are not mentioned; this undermines the reliability of the report as the royalties and actual price of sand on resale vary and the field counting of unloaded and loaded trucks exiting the quarries is normally not done by the

[21] However, recently (around 2017) the Tamilnadu Government has claimed that it is no longer allowing private miners/sub-lease holders, rather, undertaking the dredging activities by itself through the Public Works Department (PWD). It is pertinent to note here that Rule 38A of the TNMMC Rules have long back granted such exclusive rights to the State against any private entity; in reality, the Rule has been inactive as the Govt allowed private participation through the method of sub-leasing until the sudden awakening amidst complaints on illegalities and excesses in the name of dredging. In Varadharajan v District Collector & Ors, the State represented by the Advocate General has admitted to the past illegalities - WP. No. 22433 of 2017 (para 31 of the judgment).

'Tamilnadu Sand Mining 2018: Story of Nexus Exposed by a Brave Journalist', SANDRP (South Asian Network on Dams, Rivers and People), May 2, 2020, 18:30 PM

https://sandrp.in/2019/02/17/tamil-nadu-sand-mining-2018-story-of-nexus-exposed-by-a-brave-journalists/

'Bring Scheme to Supply Sand as per Building Plan, PWD told', Daily Thanthi, published on Dec 5, 2019.

https://www.dtnext.in/News/TamilNadu/2019/12/05005150/1201708/Bring-scheme-to-supply-sand-as-per-building-plan-PWD-.vpf

[22] The District Survey Reports have to be prepared in pursuance of the Ministry of Environment, Forest and Climate Change, Government of India Notification No. SO 141 (E) dated 15.01.2016 and SO 190 (E) dated 20.01.2016.

authorities; if the table is based on the data self-reported by the quarry sublease holders, there are high chances that the numbers are fabricated to match those permitted by the EC and other regulations and the real volume of sand excavated may be much larger, damaging the ecology and causing great loss to the State Treasury. Though the rationale for dredging shoals to let the river flow freely and be able to reach the distributing channels is acceptable, the ground reality (as shown in the above Figures) contradicts with this motto. These conflicting realities could have been easily identified through field inquiry; on mentioning that the DSR is written through field observations and the relevant documents, the specific places visited, the personnel and time involved are not listed in the report[23]; also, there were no pictures or review or specific mention about the Mohanur and Oruvanthur quarries which are one of the major quarries in the district. Without such data, the team has certified that "...all the safety measures will be strictly followed to prevent occupational risk during excavation, loading and transportation..."[24] How can a reporting team vouch for the future activities of the PWD? Also, the report has misstated that, "...since the operation is carrying out by the Public Works Department they properly identify the aggradation area over the river bed in the district..." But, the PWD had been subleasing the sand quarries to private people until the past 2 to 3 years which the report failed to mention. All these show that this report cannot be considered comprehensive nor neutral, especially when the State proposes to use this report as basis for future application for ECs and preparation of reports and appraisal of projects in its own Preface.[25]

On the other hand, the Central Government guidelines are uniform under a single body, for example, the 2016 Sustainable Mining Guidelines issued by the Ministry of Environment, Forests and Climate Change (MoEFCC); however, they are soft laws which act only as recommendations and the final say in their incorporation lies with the States themselves. Some of the laudable inclusions in the 2016 guidelines are State-specific suggestions have been given, for example, in Tamilnadu, it has suggested controlling within the time frame of 6 a.m. to 7 p.m. and to allow dredging of excess sand deposits in floodplains only when it is necessary to protect and maintain the groundwater table and the suggestion to decentralise SEIAA to expedite EC process has to be given to states like Uttar Pradesh and to avoid mechanical mining in Sikkim, etc. Apart from these specific suggestions, general guidelines to mining have also been provided in the Rules like clear demarcation of mining area and geo-referencing before beginning the operations, not mining below a 3-m limit at any point, preferring inactive than active water channels, accounting for the replenishment rates, etc. Recently, there is the 2020 Enforcement and Monitoring

[23] 'District Survey Report for Sand Mining - Namakkal District', (2019), April 25, 2020, 14:20 PM https://cdn.s3waas.gov.in/s3b9228e0962a78b84f3d5d92f4faa000b/uploads/2019/06/2019062065.pdf

[24] *Supra* 12, page 74.

[25] *Supra* 12, page 2. This view is also taken by the Hon'ble National Green Tribunal (NGT) in *Anjani Kumar v State of Uttar Pradesh & Ors* [O.A. 557/2017].

Guidelines for Sand Mining[26] that has accommodated recent judicial recommendations[27]; it has new provisions like restricting mining to three-fourth of the width of river and to keep the 1/fourth part as no mining zone to protect banks.[28]

This shows that the best way for sustainable management of river sand dredging will be to have a general framework with state-specific modifications. So far, the general framework is a soft law except the EIA Notifications under the Environment Protection Act; and the State-specific solutions are left unmonitored because of little to zero active Central intervention; the status quo legislations are largely coloured as they actually favour economic gain through their minimal restrictions when their proposed rationale is to protect the environment. The best way to solve this without making sand a major mineral will be passing a Central Model Law making it mandatory to incorporate its provisions by the States while reserving their right to add provisions to account for the local conditions and the right to fix royalties on the volume mined. By virtue of entries 17A, 17B, 20, 34 of List III and entries 13,42,51,54,56 and 97 of List I of seventh Schedule, it is constitutionally possible and necessary for the Central Government to make laws protecting the wildlife, forests, ecology of rivers (especially interstate rivers and valleys), fulfil the global environmental guideline consensus reached in the international fora and the Constitutional Directive Principle to protect environment under Art.48A, preventing the illegal trafficking of sand as part of interstate trade, etc. This conservation justice approach, combining the Central and State legal authority, alone will check this uncurbed mining menace throughout the country.

6 Factors to Be Accounted for in the New Framework

The general skeletal structure of a sand mining framework shall quintessentially account for all these factors:

- Authorities Granting Permits (the Departments of PWD, Geology, Hydrology, Ground Water Table Maintenance Agency, Maritime Board, Pollution Control Board should all be roped in as consulting personnel to certify the feasibility of a sand mine, its location, spatial extent, duration, etc).
- Term of licences (long-term mining leases shall be replaced by short-term mining permits or leases not extending beyond 5 years to reevaluate the continuance of mining in that area).

[26] 'Enforcement and Monitoring Guidelines for Sand Mining', Ministry of Environment, Forests and Climate Change, January 2020.

http://environmentclearance.nic.in/writereaddata/SandMiningManagementGuidelines2020.pdf

[27] *Sudarsan Das v State of West Bengal & Ors* [O.A. 173/2018] is one of the landmark judgments recently that called for revising the 2016 sand mining guidelines, laying down certain specific recommendations.

[28] *Supra* 36, page 16-17.

- EC requirements (As of the present, B2 projects with area of 5–25 ha are exempted from the process of public consultation in the EIA process; this has to be amended to allow public hearings and Gram Sabha approval to all sand mining projects; also, sand mines <5 ha must also be brought under the EIA process).
- Resale price of sand (Apart from fixing the royalties by State governments, they shall stipulate ceiling limits on the resale price of sand by leaseholders, for example, to avoid seven times increase from the actual amount paid to the State per unit load, the price ceiling on resale may be 50% or 75% of the amount paid per unit load to the State Treasury. This will ensure affordability and prevent hoarding of sand.
- Interstate movement (To avoid trafficking of sand to States where there is moratorium on mining, the concerned State Governments shall impose penalties, GPS installation and registration of trucks, their route mapping, etc).
- Community participation (For faster remediation of violations and easy & effective implementation, the States through their District administrations, form Citizen Watch groups in the quarry sites and routinely update the contact info of relevant authorities).

Apart from this, if the mining is done through leasing, there shall be holding limits on the number of mine areas held by a private person and list down the eligibility criteria for applying for the tender; inclusion of relevant nonconviction clauses will reduce mafia control of sand quarries. There should also be operational protocols as how to excavate and grade the sand, methods of disposing or replacing or utilization of the sand falling short of the construction grade during size grading. The mining rules should avoid the usage of vague terms like 'take all possible precautions for protection of environment'[29] and rather employ unambiguous legal and scientific language.

On deciding the application for opening a sand mine, criteria such as rate of sand inflow, replenishment rate of the river in that region, existence of nearby quarry clusters, last mined date if there was a quarry previously in/close to proposed area, hydraulic properties of the river soil such as its granular size, water conductivity, stability induction to the river and its banks, the geomorphology of the river basin, evapotranspiration, drainage pattern, flora and fauna population in the nearby riparian buffer, etc., shall be considered. On fixing the depth allowed to be mined, regard shall be given to the topography instead of a general mining depth limit; also, clear reference as to whether the proposed limit applies directly from the surface/only from the actual river bed after removing shoals; there shall be restrictions on the number, width and height of the bunds and clearly outlining of the mode of mining, whether manual or mechanical or semi-mechanical, after studying the regional terrain; the method of bar skimming is less detrimental to the ecologies than dry/wet pit mining (Kondolf 1997).

In this regard, the proposed 2020 EIA Notification to be passed by the Government in supersession of the 2006 Notification and its amendments have to be discussed.

[29] Rule 24 of the West Bengal Minor Mineral Concession Rules, 2016; Birsha Ohdedar (2017).

It has some beneficial provisions like reducing the size of B2 projects from 5 to 25 ha to <5 ha as they are exempted from public consultation and strict environmental impact assessments (EIA). However, it has categorized the A projects involving major and minor minerals as >100 ha which are presently >50 ha; it also pegs the B1 projects as 5–100 ha which are presently 25–50 ha. This means that all projects short of 100 ha will be mostly under the purview of SEIAA and not the MOEF; since majority of projects fall within the new B1 limit, it will enormously increase the SEIAA workload and increase the chances of manipulation by the State Governments as the SEIAA members are to be nominated by the State.[30] Also, the examination of Govt (PWD) application for mining by the SEIAA itself undermines the integrity of the EC grant[31]; the MOEF must make provisions to handle the EC application of the Govt projects by itself rather than leaving it to the State bodies.

For the projects below 5 ha, the DEAC[32] will be the appraising body whose members shall be nominated by the District Administration[33]; if there are no public hearings or strict EIA processes for the sake of convenience/smaller size of these <5-ha projects, it could lead to grave violations within that small area as the localised nature of projects and less scope for public objections could breed collusion and local corruption. The proposed notification also penalises the project proponents on failure to adhere the EC conditions and on finding any misleading false statements in the application, the project proponent may be blacklisted.[34] If the proponent is the PWD, will the same penalty applies and if so, how shall it be implemented? Because if the PWD is blacklisted, it means the mining activity cannot be carried by the government. This necessitates due revision of the proposed notification.

7 Proposed Mining Models

If the river under consideration has good water flow throughout its stretch every year and the accretions are of large quantities that they have to be removed to prevent shallowing of the river bed and flooding, the following model may be used

As shown in Fig. 17.27, the point below which, the excavation of sand will disrupt the base flow, has to be identified. For this, the mean of the base flow in the preceding 5 years shall be taken as the minimum flow to be maintained in dry seasons. To identify the volume of sand carpet necessary for maintaining that minimum flow, the granular size of the bed soil in that region, slope of the river bed, etc., shall be considered. In the first year of dredging, the accretions from the preceding years

[30] Clause 7 of the Draft EIA Notification 2020.

[31] This clearly fits the maxim ‘Nemo judex in causa sua’ (A man shall not be the judge in his own cause).

[32] ‘District Level Expert Appraisal Committee’.

[33] Clause 8 of the Draft EIA Notification 2020.

[34] Clause 17 of the Draft EIA Notification 2020.

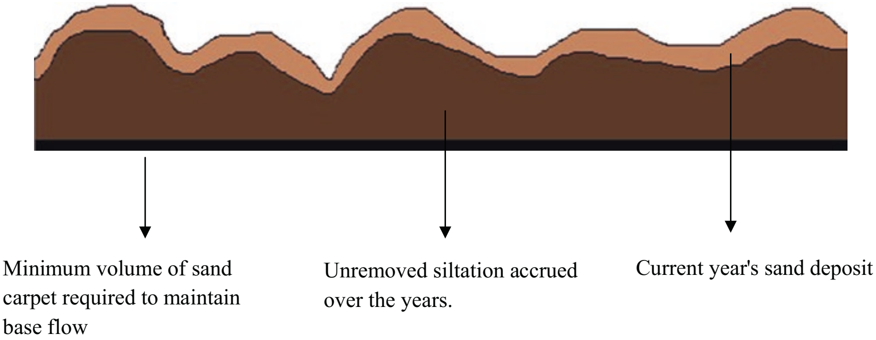

Fig. 17.27 The excavation of sand will disrupt the base flow

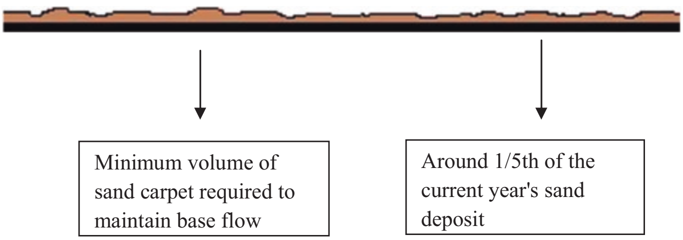

Fig. 17.28 The probable post-mining events

shall be mined fully. With regard to the present year's deposit, not more than 4/5 of the volume shall be dredged. At the end of first year's mining, the river bed will look as shown in Fig. 17.28:

From Fig. 17.28, the probable post-mining look can be seen. The reason to keep at least a minimum of 1/fifth of the current year's deposit is that, with the incoming monsoon in that year, there will be excess water flow with a fairly high velocity which can not only bring new silt but can also erode the existing carpet. If the river is to carry the fine aggregates on top of the bed as shown in Fig. 17.28, only the leftover deposition will be drifted with the minimum volume of sand to maintain base flow still intact. Since the assumption here is that the river will be depositing fair amount of silt every year requiring maintenance dredging every year, the volume to be dredged from the second year will be as follows:

If the river under consideration has slow rate of replenishment, for example, it takes 3 years to deposit enough sediment, then the dredging shall happen every 3 years and the formula will be changed to 4/5th of the 3-year sand inflow. However, only constant and close monitoring alone will bring the desired realities.

8 Alternatives to River Sand

While working on checking the uncontrolled invasive in-river mining, the alternatives to river sand in the cement mortar used in construction have to be explored. Some of the materials with potential to act as natural sand replacement are quarry dust, desert sand, foundry sand, copper slag, granulated blast furnace slag, limestone, silica, washed bottom ash, crushed and sieved construction demolition wastes, etc. (Sankh et al. 2014). Materials like incinerator ash from the incinerated solid waste have been found to exhibit greater compressive strength once they undergo pretreatment for removal of excess chloride content. Their strength, when used in cement mortar, was found to increase after 7 days of curing age and their successful replacement rate varies from 5% to 40% in the mortar; since this replacing material is the residue from the treatment of solid wastes, it leaves lesser carbon footprint than creating new cement and sand substitutes (Jun et al. 2017).

Commonly, in India, m-sand (manufactured sand) is advocated for natural sand; usually, m-sand or artificial sand is produced by rock-on-rock or rock-on-metal impact to produce fine aggregates resembling natural sand. Unlike natural sand, the m-sand will have to undergo several grading and treatment processes to become construction grade fine aggregates; however, studies show that m-sand can be effectively used as 100% replacement for ordinary sand with increased compressive and flexural strength (Elavenil et al. 2017). Since the rate of replenishment of river sand is lowering these days, to create a sustainable framework for utilising river sand for construction, alternatives must be considered. Even 30–40% replacement will conserve this depleting natural resource in this highly urbanised world.

9 Conclusion

From the discussed case studies of the two quarries in Cauvery river, it can be understood that the environmental effects of in-river sand mining are not merely theoretical. The existing conservation efforts of the State governments and their legislations in place are coloured with their desire to generate revenue and are highly nonuniform across the lengths and breadths of the Country. Realising the need for sustainable management and the preservation of not just the economy but also the riverine ecologies, buffer zones, irrigation patterns and food chain, the Central and State Governments should consciously work together under scientific guidelines to preserve this diminishing renewable source; the cost of excavated sand shall not only include the production cost but also the environmental costs as well; also, one of the current limitations observed during this study is the nonavailability of proper mining records or databases in the states; to overcome this, the governments shall use finer resolution satellite imageries to map the active and ex-sand mining areas, periodical variations in the surrounding ecologies, river course, etc., and make the database accessible to public to ensure transparency and knowledge flow. If the irreversible destruction of riverine ecosystems continues without checks in the name of maintenance dredging, the regional rivers will soon be the regional deserts with an insurmountable crisis for water the country has never seen.

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