Phosphorus fractions in the sediment of a tropical reservoir, India: Implications for pollution source identification and eutrophication

Ajoy Saha · P. K. Jesna · V. L. Ramya · S. Sibina Mol · Preetha Panikkar · M. E. Vijaykumar · U. K. Sarkar · B. K. Das

Abstract Eutrophication level in lakes and reservoirs depends on both internal and external phosphorus (P) load. Characterization of sediment P fractionation and identifying the P pollution sources are important for assessing the bio-availability of P and the dominant P source, for effectively controlling the water pollution. For determining the availability and sources of sediment P and eutrophication status, spatio-temporal variation in different P fractionation of sediment of hyper-eutrophic Krishnagiri reservoir, Tamil Nadu, India, was investigated. Sediment average total P (TP) content ranged from 4.62 to 5.64 g/kg. Main phosphorus form was the inorganic P (IP), and it makes up to 73.4–87.7% of TP. Among the different P fraction, viz. calcium bound (Ca-P), iron bound (Fe–P), aluminium bound (Al-P), exchangeable (Ex-P) and Organic-P (Org-P), Ca-P was the dominating fraction in both IP and TP. Trend of IP fraction was as follows: Ca-P > Fe–P > Al-P > Ex-P in pre-monsoon season, Fe–P > Ca-P > Al-P > Ex-P in monsoon and Ca-P > Al-P > Fe–P > Ex-P in post-monsoon. Overall the trend was as follows Ca-P > Fe–P > Al-P > Org-P > Ex-P. Bio-available-P (BAP) fractions ranged from 35.2 to 64.0% of TP, indicating its comparative higher value. Pearson's correlation matrix revealed that there was strong correlation among the different P fractions. Factor analysis indicates that different fractions of P were the dominating factor than the other sediment parameters. The observed variation in sediment P fractionation indicate the differences in source and characterization of P which is very helpful for implementation of effective management practices in controlling pollution that arises due to phosphorus in this hyper-eutrophic reservoir.

Keywords P speciation · Bio-available P · Reservoir sediment · Eutrophication · Ecological risk

A. Saha (✉) · P. K. Jesna · V. L. Ramya ·
S. S. Mol · P. Panikkar · M. E. Vijaykumar
Regional Centre of ICAR-Central Inland Fisheries
Research Institute, Bangalore 560089,
Karnataka, India
e-mail: ajoysahacob@gmail.com

U. K. Sarkar · B. K. Das
ICAR – Central Inland Fisheries Research Institute,
Barrackpore, Kolkata 700120, West Bengal, India

Introduction

To meet the ever-increasing human needs for drinking, agriculture and industries, the requirement of freshwater is increasing exponentially (Okello et al., 2015; Saha et al., 2020a). Reservoirs are major freshwater resources which provide water for drinking, irrigation, power generation, recreational purposes along with fisheries (Ramya et al., 2021; Sarkar et al., 2017). However, with rapid urbanization, industrialization, increase in agricultural nutrient run-off and anthropogenic activities, phosphorous (P) pollution in these

aquatic bio-resources is inevitable (Saha et al., 2021a; Varol, 2020; Zeinalzadeh & Rezaei, 2017). Discharge of untreated or semi-treated sewage, human and animal waste, anthropogenic activities along with agricultural nutrient input are responsible for eutrophication of any aquatic body and thereby increasing the amount of bio-available P (Xiang & Zhou, 2011). It has been estimated that every year about 18.7–31.4 million metric tons of P is lost from lithosphere to freshwater bodies (Compton et al., 2000). Recent study reported that eutrophication is a major threat in 30–40% of global freshwater lakes and reservoirs (UNEP, 20052005).

Elevated level of P in freshwater may cause high planktonic bloom, leading to turbid water and other changes in water quality which is undesirable for both human and aquatic life (Dong et al., 2011). The sediment P release ability is important in controlling the P concentration in reservoir water. Under certain environmental conditions, sediment may play the role of both source and sink of P and thus it has vital role in influencing eutrophication of overlying water (Søndergaard et al., 2003). Hence, estimation of sediment total P content may not be adequate to determine the mobilization and bio-availability of P in any type of aquatic body (Barik et al., 2019a; Soliman et al., 2017). P release potential from reservoir sediment is dependent on the various P fractions that exist in the sediment as they are controlling the mobilization and bio-availability of P in overlying water (Dong et al., 2011).

Therefore, it is very imperative to have an in-depth knowledge about the fractions of P forms exist in reservoir sediment which may help in source identification of P pollution and eutrophication in overlying water with changing environmental condition (Soliman et al., 2017). P fractions in sediment may exist in both inorganic as well as organic forms. Inorganic phosphate like iron bound (Fe–P) and aluminium bound-P (Al-P) indicates the presence of mobile and bio-available P, while non-available forms are represented by calcium (Ca) bound –P (Dong et al., 2011; Hou et al., 2014). While exchangeable –P (Ex-P), also known as available P, is the important part of inorganic P which is readily releasable from the sediment for consumption of both plant and animal (Dan et al., 2020). P bound to organic matter is known as organic (Org)-P, which is partially bio-available depending upon the environmental condition

(Sommers et al., 1972). Interchange of different fractions of P under the influence of biotic or abiotic factors may affect their seasonal or spatial variation and thus P release potential. Extent of P pollution and their inter-releasing capacity can be determined by evaluating bio-available-P (BAP) (Wang et al., 2006). Accordingly, studies on BAP are also important as they are the major contributor to the reservoir eutrophication and their variation across the space and time is furthermore important to determine the P pollution sources.

Krishnagiri reservoir is a medium tropical reservoir situated in the semiarid climatic condition in peninsular India. The multipurpose use of this reservoir water includes drinking water, irrigation, fisheries and livestock farming as well as amusement purposes. The contaminated drainage water and untreated effluent water is continuously deposited in this reservoir (Anonymous, 2019; Viswanath, 2004). Fertilizer and pesticides run-off from intensive farming practices like olericulture, floriculture and mango cultivation in the catchment of this reservoir is another warning which adds to nutrient enrichment and subsequent hyper-eutrophication (Elangovan & Murali, 2020; Saha et al., 2021a). This also leads to internal P loading as reservoir sediment acts as sink of P due to continuous deposition of P into the reservoir. In addition, this reservoir is also likely to suffer from internal P loading due to cage aquaculture practices. Therefore, controlling both of external and internal P loading is important to restore this eutrophic reservoir. For this, sediment P characterization is the important step to be taken before implementing any restoration measures. A number of works have been carried out in Krishnagiri reservoir for spatio-temporal variation of water quality (Ambujam & Sudha, 2016; Saha et al., 2021a), phytoplankton (Prabhakar et al., 2012) and zooplankton diversity (Renuga & Ramanibai, 2010) and diversity of flora and fauna (Raja et al., 2015; Ramanibai & Jeyanthi, 2010). However, thorough spatio-temporal sediment P fractionation study for Krishnagiri reservoir sediment in recent past is not available. Arunbabu et al. (201420142014) studied the sedimentation and internal phosphorus loads in Krishnagiri reservoir before the cage culture activities started in this reservoir. For understanding the mobility and bio-availability of phosphorus, it is important to evaluate various sediment P forms using different chemical extractants in a precise manner. With this,

the aim of the present endeavour was (1) to evaluate various chemical forms of sediment P and their spatio-temporal variation (2) to identify the interrelationship among various P forms along with sediment physico-chemical properties (3) to study the bio-availability of P for determining the potential P risk. This study will provide an important report on the identification of P sources and their ecological risk of a tropical hyper-eutrophic reservoir in India to minimize the eutroph-ication level.

Materials and methods

Study area

Krishnagiri reservoir (latitude of 12° 28′ N and the longitude of 78° 11′ E) was constructed across the Ponnaiyar River, Krishnagiri, Tamil Nadu, India. The Gross storage capacity of the reservoir is 68.2 MCM (Mohanakrishnan, 1988). This reservoir is one of the earliest projects, constructed in post-independence India in the dry and barren areas of the state and since then, the reservoir serves as the life line for the region and serving multiple uses of water from irrigation to fish culture (Mohanakrishnan, 1988). The area is characterized by tropical climate (Sasitharan & Elan-govan, 2020) and the surrounding area is heavily populated and industrialized. The length of the dam is 1000 m and at FRL, the reservoir has a water spread area of 12.32 km^2 and height of 22.8 m from the river bed. It has a catchment area of 5,428.43 km^2 and the catchment receives most of the urban waste and domestic effluents from the Bangalore city, the state capital of Karnataka, India. With rapid development of agricultural activities in the drainage basin and continuous deposition of sewage, the water quality of this reservoir is deteriorating constantly with nutrient enrichment that increases the primary pro-ductivity leading to hyper-eutrophication (Elangovan & Murali, 2020; Saha et al., 2021a).

Sample collection and processing

For understanding the physical nature and uniqueness of the reservoir, the reservoir was divided into four different zones (Fig. 1.), called (1) riverine zone, the entry point (inflow point) of river into the reservoir where water depth is comparatively lower with

aggregation of aquatic weeds and macrophytes; (2) transition zone, the largest area between upstream and downstream of reservoir, includes the cage aquacul-ture activity sector; (3) backwater zone, the northern part of the reservoir which seems stuck in a rut, unmoved by current event; and (4) lacustrine zone located across the dam of the reservoir with higher depth and sharp slope. Based on the elevation, water flow, accessibility, safety, latent pollution sources and sewage disposal activities, etc. sediment samples were collected from sum total of 11 sampling sites covering riverine (K10 and K11), transition (K6, K7, K8 and K9), backwater (K3, K4 and K5) and lacustrine (K1 and K2) zone. Sample collection was carried out at each sampling site for three different seasons, viz. pre-monsoon (PRM) (May), monsoon (MON) (Septem-ber) and the post-monsoon (POM) (December) season of the year 2018.

Sediment samples from 11 sites were collected with the help of manually operated grab sampler, trans-ported to the laboratory, air-dried and ground. The ground sediments were then passed through 2 mm sieve for physico-chemical parameters. Sub-surface (0.3 m) water samples from same sites were also collected through water sampler and stored immedi-ately in ice box before transported to laboratory.

In situ measurement of water pH and dissolved oxygen (DO) was taken by using portable Multipa-rameter Water Quality Instrument (YSI Professional Plus). TDS (Total dissolved solids) was determined by gravimetric analysis. Phosphate-P (PO_4^{3-}-P) and total P (TP) were analysed by UV–Vis spectrophotometer. The sampling, storage/preservation and analysis were carried out as per the APHA, (2005) guidelines. In order to determine the sediment grain texture, sedi-ments were analysed by Laser Diffraction Particle Size Analyzer (Beckman Coulter, USA, LS 13,320). Percentage of finer fraction like clay (< 2 μm) and silt (2–63 μm) as well as coarser fraction like sand (> 63 μm) were calculated. Sediment organic carbon (OC) was determined by the Walkley–Black method (Nelson & Sommers, 1996). pH and electrical con-ductivity (EC) of sediment were measured by using pH-EC meter by suspending the sediment in distilled water (1:2.5) (Biswas & Mukherjee, 1988). The titrimetric method was used for analysis of sediment calcium carbonate (% CaCO$_3$) content (Jackson, 1967).
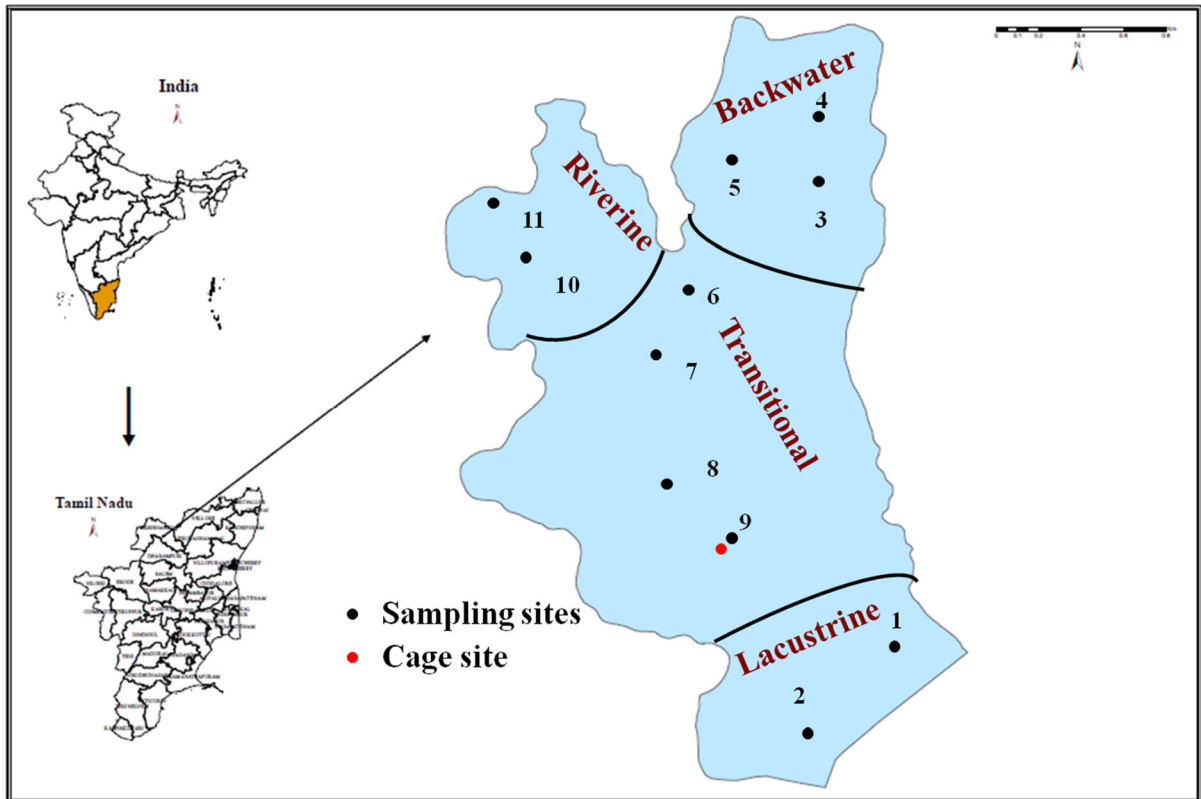
Fig. 1 Location map of the study reservoir

Analysis of sediment phosphorus fractions

Different P fractions in sediment samples were estimated by sequential fractionation following the methods of Jackson (1973) by using different chemical extractants as modified by Samanta et al. (2015). For this, sediment samples (2 g) were sequentially extracted with different extractants, viz. NH_4Cl (1.0 N), NH_4F (0.1 M), NaOH (0.1 M), and finally H_2SO_4 (0.1 M) for extraction of Ex-P, Al-P, Fe–P and Ca-P, respectively. For the extraction of each fraction, samples were shaken for 2 h and then kept overnight to settle down the sediment particles. The supernatant was decanted, and sediment samples were re-extracted with next extractant. The molybdate blue method was used to analyse the P in the supernatant (Murphy & Riley, 1962). Total-P was estimated by keeping 0.2 g of sample in H_2O_2 (40 ml; 30%) for overnight followed by digestion in tri-acid mixture (HNO_3:-$HClO_4$:H_2SO_4 at 10:4:1 ratio) and then dissolved in water before estimating spectrophotometrically. Sum total of all inorganic bound phosphorus give the total

inorganic P (TIP). Difference between total P and TIP gives the Org-P (Barik et al., 2019a).

Evaluation of sedimentary total P enrichment

In order to determine the P pollution level, sediment TP content can be taken into consideration for assessment of sediment P enrichment (Barik et al., 2019a; Dan et al., 2020). For assessing the P enrichment of Krishnagiri reservoir with reference to the sedimentary TP load, single-factor standard TP enrichment index (PEI) was calculated using the following equation,

$$PEI = C_i/C_s$$

C_i represents the sediment TP content (mg/kg) of ith sample and C_s is the standard TP content (600 mg/kg) which can cause potential ecological risk (Barik et al., 2019a) as per the criteria set by Department of Environment and Energy (DOEE), Canada (Mudroch & Azcue, 1995). Here, the standard value for TP is set as 600 mg/kg and any sediment TP value greater than

this indicates the potential ecological risk by the sediment TP content (Barik et al., 2019a; Zhang et al., 2016).

Statistical methods

Triplicate measurement was taken for all the samples and mean values are represented. Multivariate techniques such as correlation and factor analysis were used for analysis of sediment quality parameters. Pearson's correlation analysis was carried out to determine the relationship of various sediment physico-chemical properties and sediment P fractions. Principal component analysis/factor analysis (PCA/FA) was applied for potential P source identification. SPSS 20 software package was used for data analysis.

Results and discussion

Limno-chemistry of overlying water

Spatio-temporal variation in limno-chemistry of overlying water of Krishnagiri reservoir is depicted in Table 1 and Fig. 2. Spatially TP and PO_4^{3-}-P distribution in water was almost similar. The highest value for PO_4^{3-}-P was at sampling station at riverine zone as it is the inflow point of reservoir, characterized by continuous inflow of sewage water. Along with this, the shallow depth also might have caused the release of sediment P to water by wind action (Chandran & Ramamoorthi, 1984). In general, mean TP value of the Krishnagiri water was 3.76 mg/l which is greater than most of the Indian reservoirs and lakes

as reported by Upadhyay et al. (2012), for Upper lake Bhopal, India and by Saluja & Garg (2017) for Bhindawas Lake, Haryana, India. TDS was higher in the riverine zone which might be due the continuous inflow of sewage water. TDS content was lowest in MON attributed to dilution by rain water and increased during POM, but with highest values during PRM due to temperature rise. High TDS facilitates the formation and precipitation of calcium-bound P (Barik et al., 2016) and this has been also confirmed in our subsequent section. The concentration of DO varied from 3.29–13.5, 4.9–12.1 and 7.91–13 mg/l in PRM, MON and POM, respectively (Table 1), and seasonal mean DO value was non-significant to each other. However, mean DO in PRM season was higher as compared to other seasons. High algal mat formation during summer season and stimulation of phytoplankton and macrophytes to produce more oxygen as a by-product of photosynthesis resulted in high DO value during PRM. Very low DO was found at riverine zone as this zone is characterized by the inflow of water, clearly showed the impact of inflow water from the catchment area. During the whole study period, a low depth range was recorded in Krishnagiri reservoir. Spatio-temporal variations for depth values were also noticed, which fluctuated from 0.6 m at riverine zone to 8 m at lacustrine zone (Table 1). Seasonally the average depth ranged from 3.1 to 3.8 m (Table 1). Continuous siltation may be the reason for this shallow depth which also led to shrinkage of the reservoir shoreline. Shallow depth range is also an indicator for higher trophic level. Overall, the pH values ranged from 7.42 to 11.4 (Table 1), indicating that the system is alkaline in nature. Seasonal pH variation was in the

Table 1 Seasonal variation in physico-chemical characteristics of Krishnagiri reservoir water [Range (Mean $\pm$ SD)]

Season	PRM	MON	POM
Depth (m)	0.6–7 (3.8 $\pm$ 1.9[a])	0.8–7 (3.1 $\pm$ 2[c])	1.5–8 (3.7 $\pm$ 1.8[b])
pH	9.17–11.4 (10.1 $\pm$ 0.3[a])	9.15–10.6 (9.8 $\pm$ 0.5[a])	7.42–8.49 (8.0 $\pm$ 0.3[b])
DO (mg/l)	3.29–13.5 (9.9 $\pm$ 3.08[a])	4.9–12.1 (8.8 $\pm$ 2.9[a])	7.91–13.5 (9.8 $\pm$ 2.9[a])
TDS (mg/l)	812.7–831.5 (824.1 $\pm$ 6.5[a])	644.5–757.1 (721.8 $\pm$ 39.2[c])	759.8–873.7 (780.6 $\pm$ 33.9[b])
PO_4^{3-}-P (mg/l)	1.51–2.45 (1.82 $\pm$ 0.31[a])	1.5–1.9 (1.66 $\pm$ 0.5[a])	1.2–2.1 (1.6 $\pm$ 0.33[a])
TP (mg/l)	3.5–6 (4.44 $\pm$ 0.97[a])	2.5–5.1 (3.9 $\pm$ 0.74[a])	2.2–3.7(2.98 $\pm$ 0.44[b])

PRM: Pre-monsoon; MON: Monsoon; POM: Post-monsoon

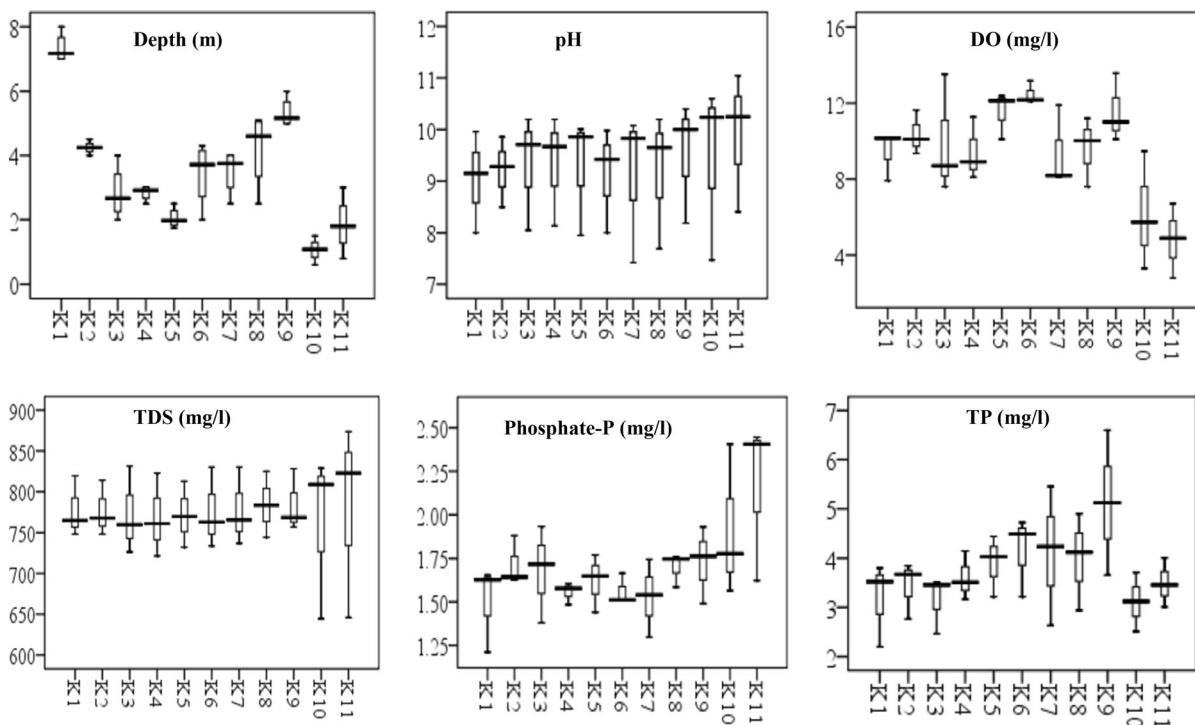Mean values bearing different superscripts under each row vary significantly (P < 0.05)

Fig. 2 Physico-chemical distribution characteristics of the overlying water varied across space

order of PRM (9.17–11.4) > MON (9.15–10.6) > POM (7.42–8.49). Spatially, riverine zone showed highest pH value.

Sediment physico-chemistry

Grain size distribution of Krishnagiri reservoir sediment displays spatial variation, while seasonal variations were almost absent (Fig. 3). In PRM, fraction of sand, silt and clay content varied from 7.77 to 70.0% (avg, 44.01 ± 19.29), 27.1 to 83.2% (avg,

50.99 ± 17.49) and 2.75 to 9.03% (avg, 5.00 ± 1.84), respectively (Fig. 3). In MON season, sand content varied from 8.1 to 67.8% (avg, 41.45 ± 17.88%), while silt and clay content varied from 31.1 to 83.8% (avg, 53.55 ± 15.67%) and 1.1 to 11.6% (avg, 5.01 ± 3.36), respectively. In POM, sand, silt and clay content varied from 9.1 to 65.8% (42.85 ± 16.53), 31.8 to 82.6% (51.85 ± 14.29) and 1.6 to 8.8% (5.30 ± 2.55), respectively. Station K2, K8, K9, K10 and K11 were dominated by coarser fraction while other stations were dominated with fine
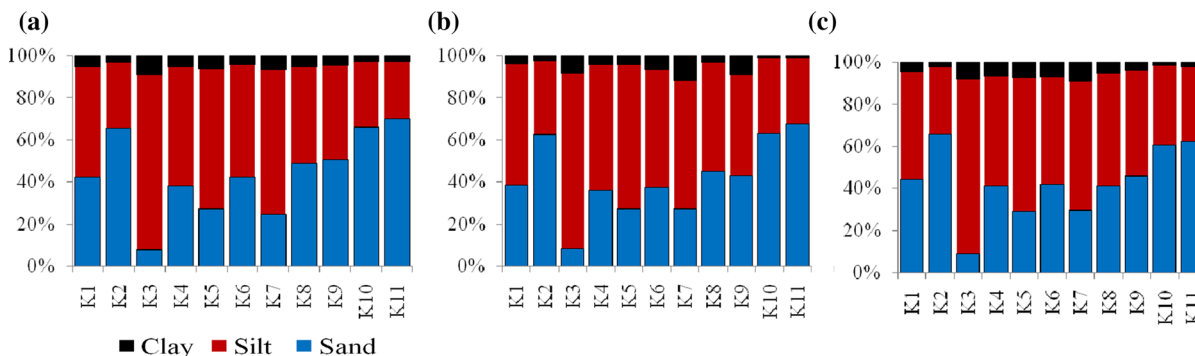

Fig. 3 Contribution of sand, silt and clay (a: pre-monsoon, b: monsoon; c: post-monsoon) in the sediment of Krishnagiri reservoir

fractions. According to textural classification of USDA (United State Department of Agriculture), sediment compositions of Krishnagiri reservoir is given in Table 2. Sediment of stations K2, K8, K9, K10 and K11 were sandy loam in texture. Sediments at K1, K4, K5, K6 and K7 were classified as silty loam, whereas station K3 showed silty texture. But seasonal variations were almost absent except few cases. Little higher content of silt and clay observed in MON and POM season might be due high flow of riverine and terrestrial water loaded with fine particles. Riverine and transitional zone were dominated by sand whereas lacustrine and backwater zones were dominated by silt. Comparatively higher sand fractions at riverine zone may be due to inflow of water which prevents the settlement of finer particles (Barik et al., 2016; Sudha & Ambujam, 2013). Except riverine zone, other parts have higher concentration of silt attributing to the transport and settlement of riverine input to the bottom of the other zones of reservoir.

　　Krishnagiri reservoir sediment was highly alkaline in nature and pH ranged from 8.38–9.1 (8.69 ± 0.24), 8.23–8.91 (8.59 ± 0.25) and 8.33–8.9 (8.62 ± 0.19) in PRM, MON and POM, correspondingly (Fig. 4). Though the spatio-temporal variation was negligible, a comparatively higher pH value was recorded at the inflow point of the reservoir. The reservoir sediment is comprised of $CaCO_3$ lithology, and sediment $CaCO_3$

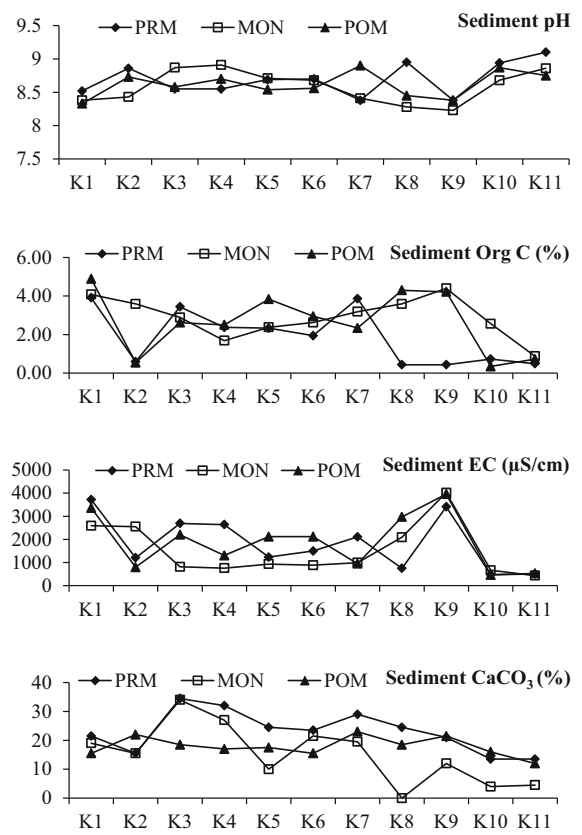**Fig. 4** Physico-chemical properties of Krishnagiri reservoir sediment (PRM: Pre-monsoon, MON: Monsoon, POM: Post-monsoon)

Table 2 USDA textural classification of Krishnagiri reservoir sediment

Sampling sites	Seasons		
	PRM	MON	POM
K1	Silt Loam	Silt Loam	Silt Loam
K2	Sandy Loam	Sandy Loam	Sandy Loam
K3	Silt	Silt	Silt
K4	Silt Loam	Silt Loam	Silt Loam
K5	Silt Loam	Silt Loam	Silt Loam
K6	Silt Loam	Silt Loam	Silt Loam
K7	Silt Loam	Silt Loam	Silt Loam
K8	Sandy Loam	Silt Loam	Silt Loam
K9	Sandy Loam	Loam	Loam
K10	Sandy Loam	Sandy Loam	Sandy Loam
K11	Sandy Loam	Sandy Loam	Sandy Loam

PRM: Pre-monsoon; MON: Monsoon; POM: Post-monsoon

content ranged from 0–34% (15.2 ± 4.5), 12–23% (17.9 ± 12) and 13.5–34.5% (23.0 ± 13.5) during PRM, MON and POM, correspondingly (Fig. 4). Higher amount of $CaCO_3$ content in the transitional zone attributing to the huge influx of weathered material brought by the river and then depositing at this interface. Backwater zone was also having high amount of $CaCO_3$ content. The reason may be that this sector comprised more of finer fraction (silt + clay) which again reaffirmed by the positive association of silt and clay with the $CaCO_3$. OC ranged from 0.43–3.91% (1.87 ± 1.42), 0.88–4.40% (2.90 ± 1.03) and 0.35–4.90% (2.66 ± 1.59) in the PRM, MON and POM, respectively (Fig. 4). High organic carbon content (up to 5%) in the sediment indicates the degree of pollution in the aquatic environment which mainly comes from domestic and municipal waste as well as from agricultural run-off. Comparatively, high OC content in the transitional

zone may be due to the cage culture activities where unutilized feeds and other organic deposited at the bottom (Parakkandi et al., 2021). Higher OC content in K1 station may be due the higher depth where influence of seasonal turbulence is negligible. Due to sandy nature, stations at riverine zones were unable to accumulate organic matter and resulting in low OC content. Higher values for sediment EC (μS/cm) were found in entire reservoir except riverine zone. EC values were higher in POM season [465–3960 μS/cm (avg, 1888 ± 1189 μS/cm)] as compared to MON [433–4020 μS/cm (1520 ± 1132μS/cm)] and PRM [478–3720 μS/cm (1839 ± 1147 μS/cm)] season (Fig. 4). In general, the OC content in bottom sediment was associated with sediment EC. Low OC accumulation at riverine zone may lead to low EC.

Total P

Spatio-temporal variation of total phosphorus (TP) content is shown in Fig. 5. In general, TP content in Krishnagiri reservoir was much higher with values varying from 2.35 to 8.8 g/kg, indicating severe P pollution. This also indicates that the sediment has the potential to release P into the overlying water and also signifies the variation in spatial distribution. The values were remarkably higher than the reference toxicity value (600 mg/kg) as defined by Ontario Ministry of Environment, Toronto, Canada (Persaud et al., 1993). TP content in bottom sediment varied from 3.0 to 8.8 g/kg in PRM, 2.35 to 6.18 g/kg in MON and 3.27 to 7.42 g/kg in POM. The TP concentration remained highest in PRM (avg, 5.64 g/kg) followed by POM (avg, 5.38 g/kg) and MON (avg,
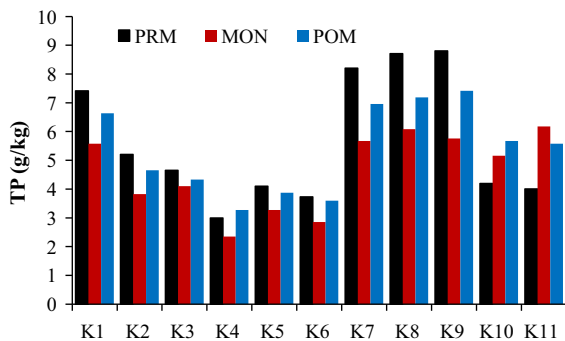
Fig. 5 Spatio-temporal variation of total P in the Krishnagiri reservoir sediment (PRM: Pre-monsoon, MON: Monsoon, POM: Post-monsoon)

4.62 g/kg) (Fig. 5). High TP content in PRM season may be due to the influx of sewage water without any dilution of the river water which resulted in concentrated accumulation of phosphorus in the sediment.

Comparatively high TP value in POM may be due to accumulation of run-off water in MON season due to rainfall run-off and irrigation run-off from agricultural field with high P concentration. P distribution in bottom sediment was influenced by the loading from both allochthonous and autochthonous source in addition to the sediment properties (Zhou et al., 2007). High TP content was found in station K8 and K9 in PRM, K8 and K11 in MON and K8 and K9 in POM, which was largely associated with cage culture activities, discharge of sewage and anthropogenic activities. Role of internal P loading might have also played an important role in defining the sediment total P content. High concentration of TP was also observed at K1 site which may be due to the high depth at this site leading to higher accumulation of phosphorus, whereas low TP content in backwater zone may due to the shallow depth. TP concentration does not represent the presence of reactive P in sediment; rather it is a miniature representation of lesser defined geochemical determinations (Barik et al., 2016).

Calcium P

Several factors like sediment type, OC content, water depth, redox condition, Ca, Fe and Al content and even the mineralization rate control the distribution of different fraction of P. Distribution of different fractions of P in Krishnagiri reservoir sediment is given in Fig. 6. It is worthy to mention that all fraction of P is not bio-available. However inorganic P can be considered as a major source of bio-available P particularly in eutrophic reservoir (Hou et al., 2014). Ca-P in sediment is mainly composed of detritus carbonate bound P, biogenic P and authigenic apatite-bound P (Barik et al., 2016).

In Krishnagiri reservoir sediment, Ca-P was found to be the most dominant fraction of inorganic P accounting for approximately 38.7, 31.3 and 36.9% of the inorganic P during PRM, MON and POM, respectively (Fig. 7a, b and c) and 34.0, 23.7 and 30.89% of the Total P, correspondingly (Fig. 7d, e and f). This is because of the prevalence of alkaline condition of Krishnagiri reservoir which makes Ca-P as a most stable fraction (Barik et al., 2019a).
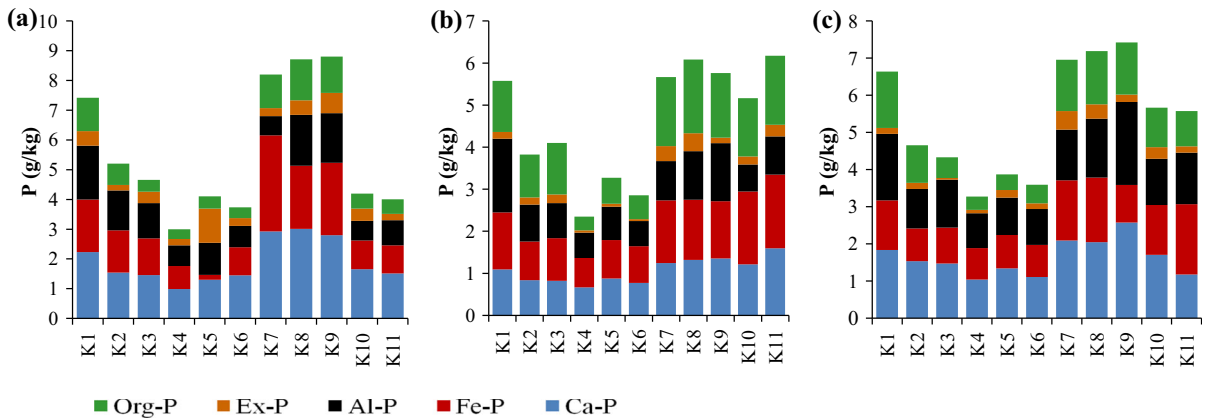
Fig. 6 P fractionation in the sediment of Krishnagiri reservoir in **a** pre-monsoon, **b** monsoon and **c** post-monsoon season
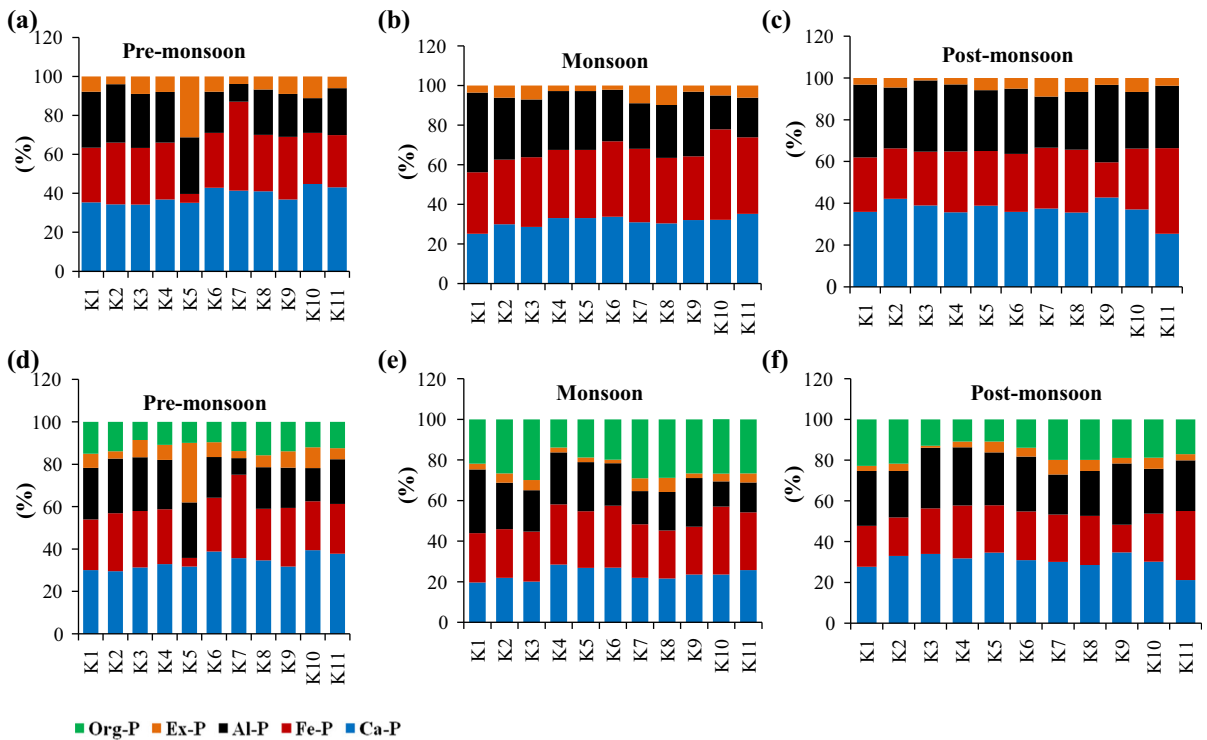

Fig. 7 P fraction contribution to inorganic P (**a**, **b**, **c**) and total P (**d**, **e**, **f**) in different sampling sites

Dominance of Ca-P has also been found in the sediment of several eutrophic reservoirs (Cheng et al., 2014; Dong et al., 2011; Kumar & Anshumali, 2019). Maximum value for Ca-P was found in PRM (avg, 1.9 ± 0.72 g/kg) followed by POM (avg, 1.63 ± 0.48 g/kg) and MON (avg, 1.07 ± 0.3 g/kg) season. High sediment pH in PRM season might have facilitated the precipitation of phosphate with CaCO₃,

led to high value for Ca-P in PRM (Anshumali & Ramanathan, 2007; Barik et al., 2019a). Furthermore, high water pH during PRM and POM may also favoured the precipitation of CaCO₃ followed by co-precipitation of phosphate in the sediment. Comparatively low pH in MON season might have broken the bonding of Ca and phosphate and thus low value of Ca-P during this season. Zooplankton grazing may be

another reason for lower value of Ca-P in MON season. Samples of riverine and transition zones showed higher value for Ca-P. K1 site of lacustrine zone also showed higher value for Ca-P. Comparatively less value for Ca-P was found in the northern sector, i.e. backwater zone of the reservoir. Spatially, the trend of Ca-P was as follows: K9 > K8 > K7 > K1 > K10 > K11 > K2 > K3 > K5 > K6 > K 4.

The water pH of riverine and transition zone was always higher which may have facilitated the precipitation of phosphate along with $CaCO_3$ resulting in the dominance of Ca-P in those sectors. The presence of Ca-P was also highly associated with the sediment Ca content. Ca-P has been reported to be dominant in several carbonate-rich shallow water bodies where phosphate gets precipitated along with carbonate (Millero et al., 2001; Renjith et al., 2011). Sediment of Krishnagiri reservoir was rich with $CaCO_3$ (up to 34.5%), which may have also contributed to this high Ca-P.

Furthermore, our previous study (Saha et al., 2021a) also revealed that inflowing water of Krishnagiri reservoir contains high level of Ca^{2+} (36.9–85.0 mg/l) which can compete with the Fe and Al and thus facilitate the formation of more Ca-P (Wen et al., 2020). Thus, eutrophication may have enhances the Ca-P formation. Though Ca-P is not readily bioavailable (Da-Peng & Yong, 20102010), but with the passage of time they may get converted into active P and thus may have the potential to increase the eutrophication level. Alkaline nature of Krishnagiri reservoir sediment indicates that calcium compounds have a dominant role in controlling the P speciation. This all may have contributed to the dominance of Ca-P over Fe-/Al-P in Krishnagiri reservoir sediment.

Iron and aluminium P

These are the redox sensitive P fraction which bound to the surface of Fe and Al of sediment (Anshumali & Ramanathan, 2007). Iron bound (Fe–P) and aluminium bound P (Al-P) are the major components of sedimentary available P which is in a constant dynamism with other fractions of sediment P. Fe–P is considered as a major part of algal available P and it indicates the source and extent of P pollution in sediment (Rose et al., 2010). However, Hedley et al. (1982) opined that due to binding of this fraction of P

with humus and oxides of Al and Fe, they are less available for plants.

Fe–P in the Krishnagiri reservoir sediment was highest in PRM (0.17 to 3.23 g/kg, avg, 1.46) followed by MON (0.7 to 1.75 g/kg, avg, 1.23) and POM (0.85 to 1.89 g/kg, avg, 1.22) (Fig. 6). However, percentage-wise Fe–P accounts for 28.2, 35.7 and 27.7% of IP during PRM, MON and POM season, respectively (Fig. 7a–c), whereas it was 24.7, 27.0 and 23.3% of TP, respectively (Fig. 7d, e and f). It indicates that dominance of Fe–P based on contribution towards IP and TP followed the order as follows MON > PRM > POM. Proportional contribution of Fe–P to total P in different eutrophic reservoirs of India and world is as follows: 1.8–8.1% in Govind Ballabh Pant Sagar reservoir, India (Kumar & Anshumali, 2019), 17.48–28.19% in Panjiakou reservoir, China (Wen et al., 2020), 27% in Gargalheiras reservoir, Brazil, and 37% in Cruzeta reservoir, Brazil (Cavalcante et al., 2018). The increase in pH during PRM and POM leads to mobilization of P from Fe–P as phosphate ions is replaced by hydroxide ion in highly alkaline condition (Anshumali & Ramanathan, 2007). This resulted in low % of Fe–P in PRM and POM than MON. The spatial variation indicates that K7 and K9 site showed maximum Fe–P content in PRM and the values were maximum in site K10 and K11 during MON and K8 and K11 site during POM season. The site K10 and K11 are situated at the inflow point, where sewage from Bangalore city enters into the reservoir (Anonymous, 2019; Viswanath, 2004). Other sites are situated in the transitional zone where cage culture activities might have increased the value.

With fluctuating redox condition, P fraction in Al-P remains unstable and this fraction of P is the most fluctuating fraction depending upon the environmental condition (Zhang et al., 2019). However, it is stable in wide pH range (5.5.–9.0) (Wen et al., 2020). Similar to Fe–P, this fraction of P also indicates the sewage inflow and is useful for identifying the anthropogenic source of allochthonous P (Ruban et al., 1999). Al-P content in Krishnagiri reservoir was highest in the POM (0.94 to 2.23 g/kg, avg, 1.35) followed by PRM (0.64 to 1.81 g/kg, avg, 1.12) and MON (0.6 to 1.75 g/kg, avg, 0.95) (Fig. 6). Al-P accounting for 23.6, 27.8 and 30.6% of IP (Fig. 7a–c) and 20.7, 21.1 and 25.4% of TP (Fig. 7 d, e and f) during PRM, MON and POM, season, respectively. In all the seasons, Al-P content was maximum at riverine and transitional zone than

other sites. As already mentioned, Fe–P or Al-P is associated with pollution sources further suggesting the anthropogenic impacts.

Exchangeable-P

Exchangeable-P (Ex-P) is the indispensable fraction of sediment P, which is readily releasable from the sediment through decomposition of organic matter and dead cell or by desorption or bioturbation by sediment dwelling micro-benthos (Adhikari et al., 2015; Andrieux-Loyer & Aminot, 2001; Dan et al., 2020). Thus, they become a part of sediment–water interface and finally released into the overlying water making it available for consumption by phytoplankton, hence affecting the reservoir productivity. This fraction of sediment P is also known as available/ loosely bound/water soluble/hydrolysed/labile P and its concentration is directly related to bio-available P as well as provides estimation about the readily available P for utilization by plant and animal (Barik et al., 2016).

Water temperature, pH, salinity and DO content of reservoir water are some of the crucial factors which control the availability and the distribution of sediment Ex-P (Bastami et al., 2018; Dan et al., 2020; Yang et al., 2016). Their distribution is also influenced by the presence of other competing ions (OH^-, SO_4^{2-} and Cl^-, etc.) in the sediment adsorption site (Barik et al., 2019b; Dan et al., 2020; Yang et al., 2019). Average sediment Ex-P content in Krishnagiri reservoir was lowest among the different IP fractions (Fig. 6). Ex-P content was found to be maximum in PRM which is consistent with the highest TP content in same season. Ex-P represents < 9% of TP in the sediments, ranging from 3.86 to 8.37% of TP (Fig. 7 d, e and f). This also corroborates the previous studies (Barik et al., 2019a; Wen et al., 2020) which reported least contribution of Ex-P to total P. Samples collected from near the cage culture site and at the inflow point of reservoir showed higher value for Ex-P, indicating that distribution of Ex-P is significantly affected by both external and internal inputs. Though the Ex-P constitutes the least portion of TP, its presence is very important even at very low concentration as they determine the P-content in sediment-pore water and finally in overlying water contributing more towards reservoir eutrophication.

Organic P

P fraction bound to organic matter forms the Organic P (Org-P). High clay content along with high deposition of terrestrial organic matter results in high Org-P content (Hou et al., 2014; Zhang et al., 2012). However, according to Frossard et al. (1989), Org-P is comparatively loosely bound to sediment particles than the inorganic bound P. Sinking of the organism in the bottom sediment after their death also increases the Org-P content (Joy et al., 2019). Since Org-P are attached to the discrete mineral phase, not firmly fixed and thus may get detached easily contributing to the eutrophication in aquatic body (Hou et al., 2014). Org-P content in Krishnagiri reservoir sediment varied from 0.33 to 1.38 g/kg (avg, 0.73 ± 0.4 g/kg) in the PRM, 0.33 to 1.75 g/kg (avg, 1.17 ± 0.49 g/kg) in the MON and 0.36 to 1.51 g/kg (avg, 0.96 ± 0.44 g/kg) in the POM (Fig. 6). Compared to IP, Org-P showed lower value and comprised of 8.5–15.8% (avg, 12.34%), 13.9–29.9% (avg, 24.43%) and 10.9–22.8% (avg, 17.05%) of TP in PRM, MON and POM, respectively (Fig. 7d–f). Organic P content was high in MON season, when the sediment pH was low as compared to PRM and POM season and Ca–P was also found to be lowest. In general, due to relatively high rate of mineralization in MON, lower value for Org-P is expected. However, the results were opposite which indicate that the Org-P fraction in Krishnagiri reservoir sediment seems to be resistant to the environmental changes. Surface heterogeneity indicates that sampling points at riverine and transitional zone showed maximum value for Org-P (Fig. 6).

From the present study it can be easily understood that IP was the major contributor to TP, accounting 87.66, 73.4 and 83% of TP during PRM, MON and POM, respectively. In IP, dominant fraction was the Ca-P. Among the different IP fraction, the trend was Ca-P > Fe–P > Al-P > Ex-P in PRM, while the trend was Fe–P > Ca-P > Al-P > Ex-P in MON and Ca-P > Al-P > Fe–P > Ex-P in POM season. Overall, among all the IP and Org-P, the trend was Ca-P > Fe–P > Al-P > Org-P > Ex-P.

Bio-available phosphorus distribution

Bio-available P (BAP) indicates the degree of pollution and inherent P releasing ability. BAP is sum total of instantaneously available P and latent P which by

physical, chemical or biological means can easily be transformed into active P (Wang & Liang, 2015). Reports indicate that there is a proportional relationship between BAP and readily released P from sediment which is utilized by algae and phytoplankton (Zhu et al., 2013). For calculating BAP, it is very much important to have knowledge on P fractionation of sediment (Hou et al., 2009). In general Ca-P in sediment, both autogenic and biogenic, are not soluble in freshwater and cannot play any role in P enrichment of overlying water. However, through re-mineralization process by microbes, Org-P may become bio-available (Wen et al., 2020). It is also reported that 50–60% of sediment Org-P can easily be degraded or hydrolysed to BAP (Wang & Liang, 2015), whereas Ex-P is also readily releasable from sediment and thus is available in overlying water which can be utilized by primary producers (Bastami et al., 2018). Under certain circumstances like reducing environment, Fe–P also gets converted to BAP. Hence, in this study sum total of Fe–P, Org-P and Ex-P was taken into consideration for calculating the BAP of Krishnagiri reservoir sediment which represents the amount of sediment P available for primary producer (Wen et al., 2019).

Variation of BAP across the space and time is shown in Fig. 8. Fraction of BAP varied from 41.1 to 56.5% (1.31 to 4.63 g/kg), 45.9 to 64.0% (1.08 to 3.67 g/kg) and 35.2 to 54.4% (1.3 to 3.55 g/kg) of TP in PRM, MON and POM, respectively. Narrow seasonal variation was observed for BAP. However, spatially it varied considerably (Fig. 8). The sampling sites at transitional zone showed higher BAP in the sediments, which indicates that the BAP has a close association with sediment internal P loads and cage
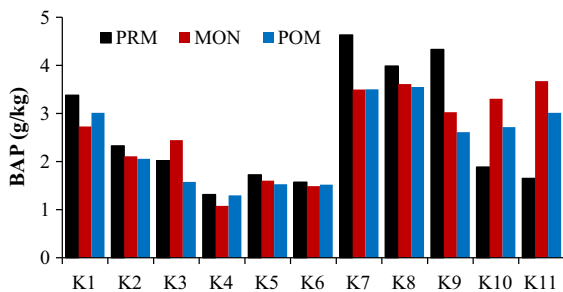

Fig. 8 Spatio-temporal variation of bio-available P in the Krishnagiri reservoir sediment (PRM: Pre-monsoon, MON: Monsoon, POM: Post-monsoon)

culture activities. Compared to other aquatic bodies in India and world (Barik et al., 2019a; Wen et al., 2020), BAP in Krishnagiri reservoir was found to be higher as this reservoir is receiving high amount of external P through heavy discharge of both agricultural and industrial effluents which make this aquatic body vulnerable to ecological risk owing to algal blooms.

Potential P source and its ecological risk in Krishnagiri reservoir

High sediment P content in Krishnagiri reservoir was due to the external P source like continuous deposition of untreated and partially treated sewage and industrial waste along with agricultural run-off. Internal P loading originates from a pool accumulated in the sediment at high external loading, is also responsible for this high value. In addition, due to cage cultural activities this reservoir also suffers from another source of internal P loading. This suggests that even after sewage diversification, eutrophication process may still persist for long-term if internal P loading due to cage culture activities is continuing in future.

Sediment P loads in Krishnagiri reservoir were comparatively higher than other eutrophic reservoirs. Anthropogenic activities, domestic sewage discharge, cage-fish culture along with release of P through fish bait and feeding resulting in high sediment internal and external P loading which leads to algal bloom (Wen et al., 2019). It has been reported that only 25–30% of the feed applied to the cage aquaculture is consumed by the fish for their growth and rest of the amount is released into the water or get accumulated into the sediment (Chen et al., 1999). Meanwhile, different point and non-point sources around this reservoir also contributing to this P loading (Ambujam & Sudha, 2016). Municipal sewage treatment plant discharges and industrial effluent treatment plant discharges (Ambujam & Sudha, 2016) is the point source of pollution for this reservoir. There are 16 villages which are benefited by this reservoir (Sudha & Ambujam, 2012) and thus this reservoir is also influenced by the human activities. Fertilizer and pesticides run-off from intensive farming practices like olericulture, floriculture and mango cultivation in the catchment of this reservoir is another warning to this aquatic body. So, sources of P pollution in this reservoir are likely to the industrial, agricultural and anthropogenic activities. Once P enters into the

reservoir, it can go through an internal cycle within food-webs or be retained in the sediment, which accumulates over time, creating P-enriched sediment (Das et al., 2012; Song et al., 2017). Furthermore, high TP content in transitional zone due to cage culture activities may increase the P content in overlying water through internal P loading. In addition, the Krishnagiri reservoir is a shallow water reservoir and high water temperature at the interface of sediment–water may likely to increase the internal sediment P releasing ability. Further, the high proportion of BAP and the shallower water depth of this reservoir may enhance water P concentrations via internal P loading driven by wind waves.

In order to determine the P load in Krishnagiri reservoir, phosphorus enrichment index (PEI) was calculated and represented by box-whisker plot (Fig. 9). The PEI ranged between 3.92 and 14.67 with an average of 8.69 ± 2.94 and values were always > 1 in all the zones of the reservoir irrespective of season. This quite high value suggests that this reservoir sediment is severely affected by anthropogenic activities and thus poses eutrophication threat. PEI values of Krishnagiri reservoir were relatively very high as compared to other eutrophic water bodies (Barik et al., 2019a; Wang et al., 2013). The sediments in the riverine and transitional zones of the reservoir showed more serious pollution than the downstream region. According to Ontario sediment quality guidelines, sediment TP content > 2000 mg/kg indicates that the sediment is severely polluted by P and may affects the sediment-dwelling organism (Persaud et al., 1993). The results of the present study also suggest that the sediment TP content may pose a severe threat for sediment-dwelling organism which calls for immediate action to control this situation.
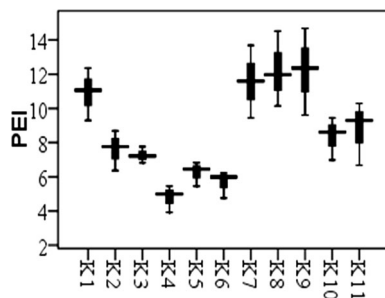

Fig. 9 Box-whisker plot showing spatial variation for phosphorus enrichment index (PEI)

In precise, the dominant sources of P pollution in Krishnagiri reservoir are both external P influx and internal P sources from internal sediment P pool and cage aquaculture activities. The degree of P pollution is affected by several factors like water depth, temperature and other disturbances the reservoir experience. For maintaining safety and security of this aquatic bio-resource, sewage diversification may be the important step. However, internal P loading due to cage culture may also increase the P content in overlying water (Yan, 2005). So, P input from cage culture activities also has to be controlled to minimize the eutrophication level in Krishnagiri reservoir.

Correlation analysis

Pearson's correlation analysis (Table 3) revealed that sediment OC and CaCO_3 is positively correlated with finer fractions like silt and clay, while it was negatively associated with sand fraction indicating that the sediment geochemistry is influenced by the grain texture of sediment. It is a well-known fact that the finer fraction (silt and clay) is rich in nutrients and organic matter than coarser fraction like sand (Renjith et al., 2011). This may be due to the formation of ionic bond with cations which lead to attachment of organic matter with finer fraction, whereas coarser particle like sand fraction in soil/sediment increases due to the oxidation of sediment organic matter (Shoja et al., 2017). Table 3 also reveals the obvious negative association between pH and OC. Researchers opined that high OC content results in higher organic acid accumulation and thus pH reduction (Saha et al., 2020b; Sposito, 2008).

Strong positive relation of TP and TIP indicates that TIP is the major contributor to TP (Table 3). Strong interrelation was also found among the different inorganic P fractions (Table 3). TIP has the most significant association with Ca-P indicating, IP was caused by Ca-P. Ex-P also showed a strong positive association with TIP, TP and Ca-P. Very weak relation between TP and OC indicates that the other factors like inorganic P emission, phosphate mining, etc. also influence the TP content (Wen et al., 2019). Correlation coefficient between Ca-P and TP was 0.886, and this value was comparatively higher than the relation of Fe–P and Al-P with TP. This indicates the dominance of autogenic Ca-P which is less influenced by the eutrophication level and can be considered as

Table 3 Correlation between different phosphorus fractions and other geochemical parameters

	Sand	Silt	Clay	pH	OC	CaCO$_3$	EC	Ex-P	Al–P	Fe–P	Ca-P	IP	Org-P	BAP	TP
Sand	1														
Silt	− 0.995**	1													
Clay	− 0.804**	0.740**	1												
pH	0.272	− 0.26	− 0.276	1											
OC	− 0.507**	0.499**	0.442*	− 0.764**	1										
CaCO$_3$	− 0.575**	0.568**	0.490**	0.062	0.079	1									
EC	− 0.236	0.226	0.244	− 0.819**	0.694**	0.145	1								
Ex-P	− 0.061	0.054	0.088	0.0	− 0.127	0.155	0.045	1							
Al– P	0.039	− 0.042	− 0.008	− 0.358*	0.237	− 0.002	0.590**	0.219	1						
Fe–P	0.101	− 0.105	− 0.05	− 0.214	− 0.012	− 0.056	0.14	0.11	0.262	1					
Ca-P	0.102	− 0.11	− 0.028	− 0.12	− 0.06	0.14	0.309	0.409*	0.567**	0.694**	1				
IP	0.087	− 0.095	− 0.022	− 0.245	0.02	0.06	0.371*	0.437*	0.681**	0.800**	0.944**	1			
Org-P	0.172	− 0.182	− 0.074	− 0.374*	0.25	− 0.350*	0.215	0.115	0.473**	0.567**	0.406*	0.563**	1		
BAP	0.128	− 0.137	− 0.045	− 0.305	0.084	− 0.166	0.195	0.341	0.430*	0.887**	0.695**	0.839**	0.838**	1	
TP	0.118	− 0.127	− 0.038	− 0.302	0.084	− 0.045	0.364*	0.391*	0.688**	0.811**	0.886**	0.975**	0.733**	0.916**	1

**Correlation is significant at the 0.01 level (2-tailed)

*Correlation is significant at the 0.05 level (2-tailed)

relatively stable fraction of sediment P (Kaiserli et al., 2002).

Poor relation between OC and Org-P implies that the Org-P must have come from residual form composed of refractory materials (Joy et al., 2019). However, TP has strong association with Org-P. Negative relation between Fe–P and Al-P with sediment pH indicates that the presence of these fractions is pH dependent and they get desorbed in acidic pH (Barik et al., 2019b). Positive association between IP and Ex-P indicates their same source dynamics. The whole thing indicates that the reservoir is enriched with P and this enrichment is attributed to anthropogenic input, domestic and industrial sewage discharge, agricultural run-off as well as cage aquaculture practices (Wen et al., 2020).

Principal component analysis (PCA)/factor analysis (FA)

PCA/FA was carried out for identifying the major factors influencing the different sediment P form. The initial set of factors was usually transformed by varimax rotation, and then, their identity was assigned. Here we have taken only those factors which represent an eigenvalue > 1. Based on the loading value, the factor loadings are considered as 'weak', 'moderate' and 'strong', when values for absolute loading is 0.50–0.30, 0.75–0.50 and > 0.75, respectively (Liu et al., 2003; Saha et al., 2021b). Four factors with eigenvalue > 1 were identified and these factors explained 84.6% of the total variance. The first factor accounts for 38.39% of total variance and controlled by total P, Ca-P, Fe–P, IP, Org-P and BAP (Table 4). It also indicated the significant association of Fe–P and BAP with TP. Factor 2 is positively associated with silt, clay and $CaCO_3$ as well as negatively associated with sand and accounts for 25.62% of total variance. Factor 3 accounts for 13.07% of total variance and its association with OC, EC and Al-P indicating that the terrestrial organic matter has a strong bonding with Al-P which minimally favoured the Al-P mobilization. Negative association of pH in factor 3 indicates that the Al-P is negatively related with sediment pH. Factor 4 accounts for 7.53% of total variance and is only influenced by a single factor, i.e. Ex-P, which indicates that the Ex-P has low influence on reservoir properties. Overall, this study signifies that the sediment P speciation have more influence on different

Table 4 Factor loadings matrix after varimax rotation

Parameters	Factor 1	Factor 2	Factor 3	Factor 4
Sand	0.074	− 0.957	− 0.172	0.033
Silt	− 0.085	0.937	0.166	− 0.034
Clay	0.008	0.857	0.167	− 0.017
pH	− 0.236	− 0.166	− 0.869	0.112
OC	0.071	0.398	0.805	− 0.247
$CaCO_3$	− 0.215	0.684	− 0.095	0.452
EC	0.063	0.120	0.911	0.263
Ex-P	0.165	0.110	− 0.102	0.643
Al-P	0.289	− 0.158	0.572	0.567
Fe–P	0.908	− 0.001	− 0.031	0.145
Ca-P	0.624	− 0.042	0.076	0.191
IP	0.749	− 0.048	0.175	0.122
Org-P	0.816	− 0.184	0.298	− 0.094
BAP	0.963	− 0.064	0.102	0.183
TP	0.836	− 0.089	0.224	0.487
Eigenvalue	5.758	3.843	1.960	1.129
% of variance	38.388	25.620	13.065	7.526
Cumulative %	38.388	64.008	77.074	84.599

forms of P than the sediment physico-chemical characteristics.

Comparison of P forms in sediment of Krishnagiri reservoir and in others lake/reservoir worldwide

The predominant phosphorus fractions in tropical and subtropical water bodies are compared in Table 5. TP content in the Krishnagiri reservoir sediment is comparable to other eutrophic water bodies from India and elsewhere (Gao et al., 2004; Jin et al., 2006; Samanta et al., 2015; Shoja et al., 2017). However, sediment TP content in the Krishnagiri reservoir was comparatively higher than the eutrophic Govind Ballabh Pant Sagar (0.16 to 0.31 g/kg) reservoir (Kumar & Anshumali, 2019) in India, Lake Dalinouer (0.493 to 0.904 g/kg) (Hou et al., 2014) in China and Utah Lake (0.28 to 1.71 g/kg) (Randall et al., 2019), in the USA. In recent years, Krishnagiri reservoir suffered nutrient enrichment due to the continuous deposition of unprocessed or partially treated sewage and industrial waste. Fertilizer and pesticides run-off from intensive farming practices like olericulture, floriculture and mango cultivation in the catchment is

Table 5 Comparison study for the concentration of P fractions (mg/kg) in lake/reservoir/wetland sediment in global

Lake/Reservoir/Wetland /locations	Range (Mean ± SD)						References
	Ex-P	Al-P	Fe-P	Ca-P	Org-P	TP	
Govind Ballabh Pant Sagar Reservoir, India	9.58–64.01	–	3.46–15.33	135.45–220.31	0.51–4.55	160.46–308.2	Kumar and Anshumali 2018
Ekbatan Dam, Iran	18.07–140.35 (53.2)	–	24.05–112.13 (51.39) (Fe- and Al-P)	36.05–310.82 (150.61)	–	520.33–3285.52 (1250.18)	Shoja et al. (2017)
Akaipur Wetland, India	0.04–4.12	29.9–190 (96.9 ± 13.4)	16.2–151 (91.0 ± 7.68)	31.1–198 (143 ± 8.42)	4890 ± 332	950–8410 (5470 ± 363)	Samanta et al. (2015)
Bhomra Wetland, India	0.04–6.21	10.4–447 (128 ± 27.0)	20.2–187 (108 ± 10.1)	385–946 (584 ± 31.2)	5060 ± 303	3200–8530 (6040 ± 344)	Samanta et al. (2015)
Taihu Lake, China	–	–	56–2476 (Fe and Al-P)	155–392	52–403	420–3408	Jin et al. (2006)
Dianchi Lake, China	–	–	–	–	–	6660	Gao et al. (2004)
Age complex Reservoir, France	3–7	29–101	147–1368	42–140	17–190	510–2290	Rapin et al. (2020)
Lake Dalinouer, China	7.92–55.31	0.78–1.61	1.27–3.95	145.37–543.21	21.3–292.7 (122.5)	493–904	Hou et al. (2014)
Gargalheiras Reservoir, Brazil	394.50	–	913.18 (Fe and Al-P)	606.62	–	487–1081	Cavalcante et al. (2018)
Cruzeta Reservoir, Brazil	345.72	–	902.30 (Fe and Al-P)	889.44	–	711–982	Cavalcante et al. (2018)
Chilika Lake, India	2.14	83.51	205.8	259.64	443.71	1003.81	Barik et al. (2019b)
Carlyle Lake, the USA	–	–	–	–	–	138–1600	Pearce et al. (2017)
Utah Lake, the USA	–	–	–	–	–	280–1710	Randall et al. (2019)

another most important warning to this reservoir as this is also contributing to the nutrient enrichment.

Like our study, some of the study showed that Ca-P is the dominant form of phosphorus fractions in the sediment of different water bodies in world (Hou et al., 2014; Kumar & Anshumalis, 2019; Shoja et al., 2017). However, Samanta et al. (2015) and Barik et al. (2019a) reported that organic P was the dominant fraction in eutrophic water bodies. Few studies (Cavalcante et al., 2018; Gao et al., 2004; Jin et al., 2006) also showed the dominance of Fe- and Al-bound P suggesting different P-release mechanism in different water bodies. Ex-P constitutes the smallest proportion of P in most of reservoir/lake sediment. Dominance of Ca in Krishnagiri reservoir sediment resulted in simultaneous adsorption and precipitation of P with Ca and thus Ca-P was dominant (Shoja et al., 2017). Furthermore, sediment pH also played a crucial role for adsorption of P (Shoja et al., 2017; Zhuan-xi et al., 2009). Changes in pH may resulted in precipitation of Fe and Al. However, as the sediment pH of Krishnagiri reservoir was always above 8.0 during the study period and thus there is a more possibility of precipitation of P with Ca rather than Fe and Al (Diaz et al., 1994). This could somewhat explain the low levels of P in the Fe and Al minerals and dominance of Ca-P.

Conclusion

The sediment P fractionation and its spatio-temporal variation in hyper-eutrophic Krishnagiri reservoir were investigated. TP content was very high in the reservoir sediment and remained highest in PRM (avg, 5.64 g/kg) followed by POM (avg, 5.38 g/kg) and MON (avg, 4.62 g/kg) season. TIP constitute 73.4–87.7% of TP and was the major contributor to sediment P, whereas Ca-P was the major contributor to TIP. Correlation analysis revealed a positive association between different fractions of IP, indicates their distribution is influenced by each other. Overall, among the all IP fractions and Org-P, the trend of dominance was as follows CA-P > Fe–P > Al-P > Org-P > Ex-P. Sediment TP content was > 2000 mg/ kg and should be concerned sufficiently according to Ontario sediment quality guidelines. In addition, a detailed study is required for determining the role of different P fractions of internal and external P loading for controlling the reservoir P load over the seasonal and spatial scale. Finally, regular monitoring of reservoir ecosystem is important to check any internal and external P loading.

References

Adhikari, P. L., White, J. R., Maiti, K., & Nguyen, N. (2015). Phosphorus speciation and sedimentary phosphorus release from the Gulf of Mexico sediments: Implication for hypoxia. *Estuarine, Coastal and Shelf Science, 164*, 77–85. https://doi.org/10.1016/j.ecss.2015.07.016

Ambujam, N. K., & Sudha, V. (2016). Environmental impacts of point and non-point source pollution in Krishnagiri reservoir: A case study in South India. *International Journal of Environmental and Ecological Engineering, 10*(4), 443–449. https://doi.org/10.5281/zenodo.1123903

Andrieux-Loyer, F., & Aminot, A. (2001). Phosphorus forms related to sediment grain size and geochemical characteristics in French coastal areas. *Estuarine, Coastal and Shelf Science, 52*(5), 617–629. https://doi.org/10.1006/ecss.2001.0766

Anonymous. (2019). District Environmental Plan Krishnagiri District. (As per Hon'ble NGT in O.A.No.710–713/2017 dated 15.07.2019). Retrieved February 26, 2021, from

https://cdn.s3waas.gov.in/
s37eacb532570ff6858afd2723755ff790/uploads/2019/11/
2019112965.pdf.

Anshumali, & Ramanathan, A. L. (2007). Phosphorus fractionation in surficial sediments of Pandoh Lake, Lesser Himalaya, Himachal Pradesh, India. *Appllied Geochemistry, 22*(9), 1860–1871. https://doi.org/10.1016/j.apgeochem.2007.03.050

APHA, AWWA, WPCF. (2005). Standard methods for the examination of water and wastewater, 21st ed. American Publication of Health Association, Washington, DC

Arunbabu, E., Ravichandran, S., & Sreeja, P. (2014). Sedimentation and internal phosphorus loads in Krishnagiri Reservoir, India. *Lakes & Reservoirs: Research & Management, 19*(3), 161–173. https://doi.org/10.1111/lre.12069

Barik, S. K., Bramha, S., Bastia, T. K., Behera, D., Kumar, M., Mohanty, P. K., et al. (2019a). Characteristics of geochemical fractions of phosphorus and its bioavailability in sediments of a largest brackish water lake, South Asia. *Ecohydrology & Hydrobiology, 19*(3), 370–382. https://doi.org/10.1016/j.ecohyd.2019.02.002

Barik, S. K., Bramha, S., Bastia, T. K., Behera, D., Mohanty, P. K., & Rath, P. (2019b). Distribution of geochemical fractions of phosphorus and its ecological risk in sediment cores of a largest brackish water lake, South Asia. *International Journal of Sediment Research, 34*(3), 251–261. https://doi.org/10.1016/j.ijsrc.2018.11.004

Barik, S. K., Bramha, S. N., Mohanty, A. K., Bastia, T. K., Behera, D., & Rath, P. (2016). Sequential extraction of different forms of phosphorus in the surface sediments of Chilika Lake. *Arabian Journal of Geosciences, 9*(2), 135. https://doi.org/10.1007/s12517-015-2217-5

Bastami, K. D., Neyestani, M. R., Raeisi, H., Shafeian, E., Baniamam, M., Shirzadi, A., et al. (2018). Bioavailability and geochemical speciation of phosphorus in surface sediments of the Southern Caspian Sea. *Marine Pollution Bulletin, 126*, 51–57. https://doi.org/10.1016/j.marpolbul.2017.10.095

Biswas, T. D., & Mukherjee, S. K. (1988). *Soil science textbook*. Total McGraw-Hill Publishing Company Limited.

Cavalcante, H., Araújo, F., Noyma, N. P., & Becker, V. (2018). Phosphorus fractionation in sediments of tropical semiarid reservoirs. *Science of the Total Environment, 619*, 1022–1029. https://doi.org/10.1016/j.scitotenv.2017.11.204

Chandran, R., & Ramamoorthi, K. (1984). Hydrobiological studies in the gradient zone of the Vellar Estuary: 1. Physico-Chemical Parameters. *Mahasagar, 17*(2), 69–77.

Chen, Y. S., Beveridge, M. C. M., & Telfer, T. C. (1999). Settling rate characteristics and nutrient content of the faeces of Atlantic salmon, *Salmo salar* L., and the implications for modelling of solid waste dispersion. *Aquaculture Research, 30*(5), 395–398. https://doi.org/10.1046/j.1365-2109.1999.00334.x

Cheng, H., Pu, X., Chen, Y., Hao, F., & Dong, L. (2014). Characterization of phosphorus species and modeling for its organic forms in eutrophic shallow lake sediments, North China. *Frontiers of Environmental Science & Engineering, 8*(6), 905–921. https://doi.org/10.1007/s11783-014-0650-4

Compton, J. S., Mallinson, D. J., & Glenn, C. R. (2000). Variations in the global phosphorus cycle. In C. R. Glenn (Ed.), *Marine authigenesis: from global to microbial* (pp. 21–33). Society for Sedimentary Geology (SEPM).

Dan, S. F., Liu, S. M., & Yang, B. (2020). Geochemical fractionation, potential bioavailability and ecological risk of phosphorus in surface sediments of the Cross River estuary system and adjacent shelf, South East Nigeria (West Africa). *Journal of Marine Systems, 201*, 103244. https://doi.org/10.1016/j.jmarsys.2019.103244

Da-Peng, L., & Yong, H. (2010). Sedimentary phosphorus fractions and bioavailability as influenced by repeated sediment resuspension. *Ecological Engeneering, 36*(7), 958–962. https://doi.org/10.1016/j.ecoleng.2010.04.014

Das, J., Daroub, S. H., Bhadha, J. H., Lang, T. A., & Josan, M. (2012). Phosphorus release and equilibrium dynamics of canal sediments within the everglades agricultural area, Florida. *Water, Air, & Soil Pollution, 223*, 2865–2879. https://doi.org/10.1007/s11270-012-1152-2

Diaz, O. A., Reddy, K. R., & Moor, J. (1994). Solubility of inorganic P in stream water as influenced by pH and calcium concentration. *Water Research, 28*(8), 1755–1763. https://doi.org/10.1016/0043-1354(94)90248-8

Dong, L., Yang, Z., & Liu, X. (2011). Phosphorus fractions, sorption characteristics, and its release in the sediments of Baiyangdian Lake, China. *Environmental Monitoring and Assessment, 179*(1–4), 335–345. https://doi.org/10.1007/s10661-010-1740-9

Elangovan, A., & Murali, V. (2020). Mapping the chlorophyll-a concentrations in hypereutrophic Krishnagiri reservoir (India) using landsat 8 operational land imager. *Lakes & Reservoirs: Research & Management, 25*(4), 377–387. https://doi.org/10.1111/lre.12346

Frossard, E., Stewart, J. W., & St. Arnaud, R. J. . (1989). Distribution and mobility of phosphorus in grassland and forest soils of Saskatchewan. *Canadian Journal of Soil Science, 69*(2), 401–416. https://doi.org/10.4141/cjss89-040

Gao, L., Yang, H., Zhou, J. M., & Lue, J. J. (2004). Lake sediments from Dianchi Lake: A phosphorus sink or source? *Pedosphere, 14*, 483–490.

Hedley, M. J., Stewart, J. W. B., & Chauhan, B. (1982). Changes in inorganic and organic soil phosphorus fractions induced by cultivation practices and by laboratory incubations 1. *Soil Science Society of America Journal, 46*(5), 970–976. https://doi.org/10.2136/sssaj1982.03615995004600050017x

Hou, D., He, J., Lü, C., Dong, S., Wang, J., Xie, Z., et al. (2014). Spatial variations and distributions of phosphorus and nitrogen in bottom sediments from a typical north-temperate lake, China. *Environmental Earth Sciences, 71*(7), 3063–3079. https://doi.org/10.1007/s12665-013-2683-6

Hou, L. J., Liu, M., Yang, Y., Ou, D. N., Lin, X., Chen, H., et al. (2009). Phosphorus speciation and availability in intertidal sediments of the Yangtze Estuary, China. *Appllied Geochemistry, 24*(1), 120–128. https://doi.org/10.1016/j.apgeochem.2008.11.008

Jackson ML (1967) Soil chemical analysis. Prentice Hall of India Pvt. Ltd, New Delhi, India.

Jackson, M. L. (1973). *Soil Chemical Analysis*. Prentice-Hall of India Pvt. Ltd.

Jin, X., Wang, S., Pang, Y., & Wu, F. C. (2006). Phosphorus fractions and the effect of pH on the phosphorus release of the sediments from different trophic areas in Taihu Lake, China. *Environmental Pollution, 139*(2), 288–295. https://doi.org/10.1016/j.envpol.2005.05.010

Joy, A., Anoop, P., Rajesh, R., Mathew, J., Mathew, A., & Gopinath, A. (2019). Spatial variation of phosphorus fractionation in the coral reef sediments of Lakshadweep Archipelago, Indian Ocean. *Chemistry and Ecology, 35*(7), 592–612. https://doi.org/10.1080/02757540.2019.1618281

Kaiserli, A., Voutsa, D., & Samara, C. (2002). Phosphorus fractionation in lake sediments–Lakes Volvi and Koronia, N. Greece. *Chemosphere, 46*(8), 1147–1155. https://doi.org/10.1016/S0045-6535(01)00242-9

Kumar, B., & Anshumali. (2019). Phosphorus fractionation in surface water and sediments of industrially polluted freshwater reservoir, India. *Chemistry and Ecology, 35*(3), 219–234. https://doi.org/10.1080/02757540.2018.1554058

Liu, C. W., Lin, K. H., & Kuo, Y. M. (2003). Application of factor analysis in the assessment of groundwater quality in a blackfoot disease area in Taiwan. *Science of the Total Environment, 313*(1–3), 77–89. https://doi.org/10.1016/S0048-9697(02)00683-6

Millero, F., Huang, F., Zhu, X., Liu, X., & Zhang, J. Z. (2001). Adsorption and desorption of phosphate on calcite and aragonite in seawater. *Aquatic Geochemistry, 7*(1), 33–56. https://doi.org/10.1023/A:1011344117092

Mohanakrishnan, S. (1988). *The Krishnagiri Reservoir Project—A Technical Document*. Anna University Publication.

Mudroch, A., & Azcue, J. M. (1995). *Manual of aquatic sediment sampling*. Lewis Publishers.

Murphy, J. A., & Riley, J. P. (1962). A modified single solution method for the determination of phosphate in natural waters. *Analytica Chimica Acta, 27*, 31–36. https://doi.org/10.1016/S0003-2670(00)88444-5

Nelson, D. W., & Sommer, L. E. (1996). Total carbon, organic carbon, and organic matter. Methods of soil analysis: Part 3 Chemical methods, Vol. 5, pp. 961–1010.

Okello, C., Tomasello, B., Greggio, N., Wambiji, N., & Antonellini, M. (2015). Impact of population growth and climate change on the freshwater resources of Lamu Island, Kenya. *Water, 7*(3), 1264–1290. https://doi.org/10.3390/w7031264

Parakkandi, J., Saha, A., Sarkar, U. K., Das, B. K., Puthiyottil, M., Muhammadali, S. A., et al. (2021). Spatial and temporal dynamics of phytoplankton in association with habitat parameters in a tropical reservoir, India. *Arabian Journal of Geosciences, 14*, 827. https://doi.org/10.1007/s12517-021-07194-0

Pearce, A. R., Chambers, L. G., & Hasenmueller, E. A. (2017). Characterizing nutrient distributions and fluxes in a eutrophic reservoir, Midwestern United States. *Science of the Total Environment, 581*, 589–600. https://doi.org/10.1016/j.scitotenv.2016.12.168

Persaud, D., Jaagumagi, R., & Hayton, A. (1993). Guidelines for the protection and management of aquatic sediment quality in Ontario. Water Resources Branch, Ontario Ministry of the Environment, Toronto.

Prabhakar, C., Saleshrani, K., Tharmaraj, K., & Kumar, V. M. (2012). Investigation of planktonic diversity in Krishnagiri reservoir, Krishnagiri district, Tamil Nadu, India. *International Journal of Pharmaceutical & Biological Archive, 3*(1), 153–156.

Raja, M., Ramkumar, R., Kavitha, M., Balasubramanian, V., Raja, R. K., Anandhi, M., et al. (2015). Ichthyofauna diversity in relation to water quality and habitat characteristics of Krishnagiri Reservoir, Tamil Nadu, India. *International Journal of Pure and Applied Biotechnology IJPAB, 2*, 1–15.

Ramanibai, R., & Jeyanthi, S. (2010). Benthic diatom diversity of Krishnagirireservoir. *Wetlands, Biodiversity and Climate Change, 22*, 1–13.

Ramya, V. L., Parakkandi, J., Saha, A., Salim, S. M., Panikkar, P., Sarkar, U. K., et al. (2021). Habitat based fish assemblage and distribution pattern in a large reservoir of peninsular India. *Arabian Journal of Geosciences, 14*, 689. https://doi.org/10.1007/s12517-021-07059-6

Randall, M. C., Carling, G. T., Dastrup, D. B., Miller, T., Nelson, S. T., Rey, K. A., et al. (2019). Sediment potentially controls in-lake phosphorus cycling and harmful cyanobacteria in shallow, eutrophic Utah Lake. *PLoS ONE, 14*(2), e0212238. https://doi.org/10.1371/journal.pone.0212238

Rapin, A., Rabiet, M., Mourier, B., Grybos, M., & Deluchat, V. (2020). Sedimentary phosphorus accumulation and distribution in the continuum of three cascade dams (Creuse River, France). *Environmental Science and Pollution Research, 27*, 6526–6539. https://doi.org/10.1007/s11356-019-07184-6

Renjith, K. R., Chandramohanakumar, N., & Joseph, M. M. (2011). Fractionation and bioavailability of phosphorus in a tropical estuary, Southwest India. *Environmental Monitoring and Assessment, 174*(1–4), 299–312. https://doi.org/10.1007/s10661-010-1458-8

Renuga, K., & Ramanibai, R. (2010). Zooplankton composition present in Krishnagiri reservoir, Tamilnadu, India. *Current Biotica, 3*, 4519–4525.

Rose, T. J., Hardiputra, B., & Rengel, Z. (2010). Wheat, canola and grain legume access to soil phosphorus fractions differs in soils with contrasting phosphorus dynamics. *Plant and Soil, 326*(1–2), 159–170. https://doi.org/10.1007/s11104-009-9990-4

Ruban, V., Brigault, S., Demare, D., & Philippe, A. M. (1999). An investigation of the origin and mobility of phosphorus in freshwater sediments from Bort-Les-Orgues Reservoir, France. *Journal of Environmental Monitoring, 1*(4), 403–407. https://doi.org/10.1039/A902269D

Saha, A., Parakkandi, J., Ramya, V. L., Salim, S. M., Panikkar, P., Vjaykumar, M. E., et al. (2021a). Evaluation of spatiotemporal variations in physico-chemical limnology, trophic status and cyanobacterial diversity of an impacted tropical reservoir, India for its sustainable management. *International Journal of Environmental Analytical Chemistry*. https://doi.org/10.1080/03067319.2021.1919654

Saha, A., Ramya, V. L., Jesna, P. K., Mol, S. S., Panikkar, P., Vijaykumar, M. E., et al. (2021b). Evaluation of spatiotemporal changes in surface water quality and their suitability for designated uses, Mettur reservoir, India. *Natural Resources Research, 30*, 1367–1394. https://doi.org/10.1007/s11053-020-09790-5

Saha, A., Salim, S. M., Sudheesan, D., Suresh, V. R., Nag, S. K., Panikkar, P., et al. (2020a). Impacts of a massive flood

event on the physico-chemistry and water quality of river Pampa in Western Ghats of India. *International Journal of Environmental Analytical Chemistry*. https://doi.org/10.1080/03067319.2020.1843026

Saha, A., Salim, S. M., Sudheesan, D., Suresh, V. R., Nag, S. K., Panikkar, P., et al. (2020b). Geochemistry, mineralogy and nutrient concentrations of sediment of River Pampa in India during a massive flood event. *Arabian Journal of Geosciences, 13*(20), 1–18. https://doi.org/10.1007/s12517-020-06053-8

Saluja, R., & Garg, J. K. (2017). Trophic state assessment of Bhindawas Lake, Haryana India. *Environmental Monitoring and Assessment, 189*(1), 32. https://doi.org/10.1007/s10661-016-5735-z

Samanta, S., Debnath, D., Maitra, N., Banerjee, M., Chowdhury, A. N., Sharma, A. P., et al. (2015). Sediment phosphorus forms and levels in two tropical floodplain wetlands. *Aquatic Ecosystem Health and Management, 18*(4), 467–474. https://doi.org/10.1080/14634988.2015.1114343

Sarkar, U. K., Sandhya, K. M., Mishal, P., Karnatak, G., Lianthuamluaia, L., Kumari, S., et al. (2017). Status, prospects, threats, and the way forward for sustainable management and enhancement of the tropical Indian reservoir fisheries: an overview. *Reviews in Fisheries Science & Aquaculture, 26*, 155–175. https://doi.org/10.1080/23308249.2017.1373744

Sasitharan, S., & Elangovan, A. (2020). Sediment and nutrient export modeling in a subwatershed of Krishnagiri resevoir project. *Pollution Research, 39*(2), 385–392.

Shoja, H., Rahimi, G., Fallah, M., & Ebrahimi, E. (2017). Investigation of phosphorus fractions and isotherm equation on the lake sediments in Ekbatan Dam (Iran). *Environmental Earth Sciences, 76*(6), 235. https://doi.org/10.1007/s12665-017-6548-2

Soliman, N. F., El Zokm, G. M., & Okbah, M. A. (2017). Evaluation of phosphorus bioavailability in El Mex Bay and Lake Mariut sediments. *International Journal of Sediment Research, 32*(3), 432–441. https://doi.org/10.1016/j.ijsrc.2017.05.006

Sommers, L. E., Harris, R. F., Williams, J. D., Armstrong, D. E., & Syers, J. K. (1972). Fractionation of organic phosphorus in Lake sediments. *Soil Science Society of America Journal, 36*(1), 51–54.

Søndergaard, M., Jensen, P. J., & Jeppesen, E. (2003). Role of sediment and internal loading of phosphorus in shallow lakes. *Hydrobiologia, 506–509*, 135–145. https://doi.org/10.1023/B:HYDR.0000008611.12704.dd

Song, K., Adams, C. J., & Burgin, A. J. (2017). Relative importance of external and internal phosphorus loadings on affecting lake water quality in agricultural landscapes. *Ecological Engineering, 108*, 482–488. https://doi.org/10.1016/j.ecoleng.2017.06.008

Sposito, G. (2008). *The Chemistry of Soils*. Oxford University Press Inc.

Sudha, V., & Ambujam, N. K. (2012). Longitudinal heterogeneity of sediment characteristics during southwest monsoon season in hyper-eutrophic Krishnagiri reservoir, India. *Environmental Monitoring and Assessment, 184*(3), 1287–1298. https://doi.org/10.1007/s10661-011-2040-8

Sudha, V., & Ambujam, N. K. (2013). Characterization of bottom sediments and phosphorus fractions in hyper-eutrophic Krishnagiri reservoir located in an agricultural watershed in India. *Environment, Development and Sustainability, 15*(2), 481–496. https://doi.org/10.1007/s10668-012-9388-y

UNEP. Millennium ecosystem assessment synthesis report. Prepublication final draft approved by MA board on 23rd march, the United Nations environment programme; 2005.

Upadhyay, R., Pandey, A. K., Upadhyay, S. K., Bassin, J. K., & Misra, S. M. (2012). Limnochemistry and nutrient dynamics in Upper Lake, Bhopal, India. *Environmental Monitoring and Assessment, 184*(11), 7065–7077. https://doi.org/10.1007/s10661-011-2480-1

Varol, M. (2020). Spatio-temporal changes in surface water quality and sediment phosphorus content of a large reservoir in Turkey. *Environmental Pollution, 259*, 113860. https://doi.org/10.1016/j.envpol.2019.113860

Viswanath, R. (2004). Mucky transfer Bangalore's sewage contaminates Tamil Nadu's water. Down to earth. April 30, New Delhi. Retrieved December 7, 2020, from https://www.downtoearth.org.in/news/mucky-transfer-11152.

Wang, L., & Liang, T. (2015). Distribution characteristics of phosphorus in the sediments and overlying water of Poyang Lake. *PLoS ONE, 10*(5), e0125859. https://doi.org/10.1371/journal.pone.0125859

Wang, L., Ye, M., Li, Q., Zou, H., & Zhou, Y. (2013). Phosphorus speciation in wetland sediment of Zhujiang (Pearl) River Estuary, China. *Chinese Geographical Science, 23*(5), 574–583. https://doi.org/10.1007/s11769-013-0627-4

Wang, Q., Jiang, X., Jin, X., & Xu, Y. (2006). Distribution of phosphorus fractions and bio-available phosphorus forms and their relationship in the sediments from different regions of Lake Taihu. *Journal of Lake Science, 18*, 120–126.

Wen, S., Wang, H., Wu, T., Yang, J., Jiang, X., & Zhong, J. (2020). Vertical profiles of phosphorus fractions in the sediment in a chain of reservoirs in North China: implications for pollution source, bioavailability, and eutrophication. *Science of the Total Environment, 704*, 135318. https://doi.org/10.1016/j.scitotenv.2019.135318

Wen, S. L., Wu, T., Yang, J., Li, X., Gong, W., & Zhong, J. (2019). Distribution characteristics and exchange flux of nitrogen and phosphorus at the sediment-water interface of Daheiting Reservoir in winter. *Journal Environmental Science (china), 39*(3), 1217–1225.

Xiang, S. L., & Zhou, W. B. (2011). Phosphorus forms and distribution in the sediments of Poyang Lake, China. *International Journal of Sediment Research, 26*(2), 230–238. https://doi.org/10.1016/S1001-6279(11)60089-9

Yan, N. D. (2005). Research needs for the management of water quality issues, particularly phosphorus and oxygen concentrations, related to salmonid cage aquaculture in Canadian freshwaters. *Environmental Reviews, 13*(1), 1–9. https://doi.org/10.1139/a05-001

Yang, B., Lan, R. Z., Lu, D. L., Dan, S. F., Kang, Z. J., Jiang, Q. C., et al. (2019). Phosphorus biogeochemical cycling in intertidal surface sediments from the Maowei Sea in the northern Beibu Gulf. *Regional Studies in Marine Science, 28*, 100624. https://doi.org/10.1016/j.rsma.2019.100624

Yang, B., Liu, S. M., Wu, Y., & Zhang, J. (2016). Phosphorus speciation and availability in sediments off the eastern

coast of Hainan Island, South China Sea. *Continental Shelf Research, 118*, 111–127. https://doi.org/10.1016/j.csr.2016.03.003

Zeinalzadeh, K., & Rezaei, E. (2017). Determining spatial and temporal changes of surface water quality using principal component analysis. *Journal of Hydrology: Regional Studies, 13*, 1–10. https://doi.org/10.1016/j.ejrh.2017.07.002

Zhang, R., Yin, A., & Gao, C. (2019). Sediment phosphorus fraction and release potential in the major inflow rivers of Chaohu Lake, Eastern China. *Environmental Earth Sciences, 78*(4), 117. https://doi.org/10.1007/s12665-019-8086-6

Zhang, Y., Gao, X., Wang, C., Chen, C. T., Zhou, F., & Yang, Y. (2016). Geochemistry of phosphorus in sediment cores from Sishili Bay, China. *Marine Pollution Bulletin, 113*(1–2), 552–558. https://doi.org/10.1016/j.marpolbul.2016.08.050

Zhang, Z. B., Tan, X. B., Wei, L. L., Yu, S. M., & Wu, D. J. (2012). Comparison between the lower Nansi Lake and its inflow rivers in sedimentary phosphorus fractions and phosphorus adsorption characteristics. *Environmental*

Earth Sciences, 66(5), 1569–1576. https://doi.org/10.1007/s12665-011-1400-6

Zhou, J., Wu, Y., Kang, Q., & Zhang, J. (2007). Spatial variations of carbon, nitrogen, phosphorous and sulfur in the salt marsh sediments of the Yangtze Estuary in China. *Estuarine, Coastal and Shelf Science, 71*(1–2), 47–59. https://doi.org/10.1016/j.ecss.2006.08.012

Zhu, M., Zhu, G., Zhao, L., Yao, X., Zhang, Y., Gao, G., et al. (2013). Influence of algal bloom degradation on nutrient release at the sediment–water interface in Lake Taihu, China. *Environmental Science and Pollution Research, 20*(3), 1803–1811. https://doi.org/10.1007/s11356-012-1084-9

Zhuan-xi, L., Tang, J. L., & Wang, T. (2009). P retention capacity of agricultural headwater ditch sediments under alkaline condition in purple soils area. *China Ecological Engineering, 35*, 57–64. https://doi.org/10.1016/j.ecoleng.2008.09.02