Seasonal variations in water quality of a tropical reservoir: considerations for cage aquaculture expansion

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Abstract. A study of water quality of the 30 year-old Batang Ai hydroelectric reservoir at different depths and locations was conducted to give a better understanding of the water quality status for aquaculture expansion. The study was conducted at four locations where two were identified as potential cage culture sites but nearby current operations. Temperature and dissolved oxygen (DO) profilings and 13 other parameters at 0.2, 10 and 20 meters depths were also studied. It was found that the difference in DO between seasons were significant. Pockets of high nutrients and hydrogen sulphide were observed at 10 and 20 m nearby the two cage culture sites in dry season. Additionally, regardless of season, those sites showed significantly higher 10 m and 20 m suspended solids. At one of the locations near cage culture, dry season total phosphorus and ammonia were the highest at all depths and chlorophyll-*a* was at eutrophic level at 10 m depth. All these are attributed to aquaculture waste, pre-marketing fish processing and the lack of mixing. Water Quality Index shows that the 0.2 m depth is suitable for sensitive aquatic organism. However, dry season 10 m water column of the two potential cage culture sites was polluted. Though the cages do not normally extend to 10 m deep, when overturn occurs, it could cause fish kills. Thus, in reservoir cage culture, elimination of reservoir pre-market processing is crucial and considerations have to be made on the proximity to the current operations and dry season water quality limitations.

Key Words: aquaculture, thermocline, nutrients, oxycline, sulphide, trophic state.

Introduction. Tropical reservoirs are suitable places for fish cage culture especially in developing countries where it can alleviate poverty through increased and low cost food production (Kaggwa et al 2011). Those created by hydroelectric dams have large surface area and volume of water and thus, are ideal locations for inland cage aquaculture as an alternative livelihood for the displaced people and an effective non-water-consumptive secondary use of the reservoir resources (De Silva & Phillips 2007). The 30 year-old Batang Ai Reservoir located in the rural area of Sarawak state on Borneo Island is without exception. It has been an important source of freshwater fish since wild fish from inland rivers is not able to meet the demand. The Batang Ai Reservoir, 85 km^2 in surface area, was created when the dam was built in 1984. With the support of the Sarawak Government, cage culture began in 1993 (Pusin 1995) with a production of 3 metric tons. From 1995 to 2010, the annual production fluctuated between 142 and 449 metric tons and the maximum reached 744 metric tons in 2011 with *Oreochromis* sp. being the dominant species cultured. During those 15 years, local owners' production totalled 1.7% and it contributed to the rural livelihood. Subsequently, the Sarawak Land Consolidation and Rehabilitation (SALCRA) was appointed by the State government as the lead agency to implement the Batang Ai Integrated Fish Cage Culture Project which is a large-scale, technology-focused, sustainable and integrated freshwater aquaculture project aiming to produce high value and quality fresh tilapia fish for both the local and international markets in order to create new commercial fish farmers among the local population since the current production is still below the carrying capacity (SALCRA 2012).

Although cage culture expansion is a good initiative, for the cage culture industry and reservoir fisheries to be sustainable, quality water environment in the reservoir has to be maintained. Environmental impact of cage culture on the water bodies includes deteriorating water quality mainly due to nutrients enrichment from waste feeds, faeces and urine as reported in different parts of the world such as Brazil, China and Indonesia (Beveridge 1984; Starling et al 2002; Guo & Li 2003; Abery et al 2005; Effendie et al 2005; Hayami et al 2008; Zhou et al 2011). In extreme cases, fish kills occurred such as those in Saguling, Cirata and Jatiluhur reservoirs (Abery et al 2005). Nutrients from aquaculture has been cited as an in-lake problem (Sharip & Zakaria 2008) and dry season trophic state assessment of 15 lakes and reservoirs in West Malaysia indicated that all are eutrophic (Sharip et al 2014). In Batang Ai reservoir in Malaysia, limited literature showed that the impact of suspended solids and biochemical oxygen demand could be observed up to 100 m from the culture sites (Nyanti et al 2012). Although studies of water quality have been conducted in Batang Ai reservoir (Nyanti et al 2012; Ling et al 2012b; Ling et al 2013a) they were at different locations in the reservoir. New studies were needed as water quality may have changed with the addition of fish cages by the operators and with the change in the management of harvested fish for the market. In addition, new sites have been identified as potential new fish cage culture development in Batang Ai Reservoir, thus, water quality at those sites had to be determined. Therefore, the objective of this study was to determine the water quality at three different depths at the two potential aquaculture sites which were also nearby active cage culture sites, an abandoned cage culture with new active cages compared with the confluence of Batang Ai River and Engkari River during dry season and wet season.

Material and Method. This study was conducted in Batang Ai Hydroelectric Dam Reservoir, which is located in Lubok Antu, Sarawak, about 250 km south-east of Kuching. The reservoir has a catchment area of 1200 km^2, water surface area of 85 km^2, gross storage volume of about 2870 million cubic meters and dead storage of about 1630 million cubic meters (SIWR 2008). The depth of the reservoir ranged from 14 m to 63 m (Ling et al 2012b). Water samples were collected at four stations (Figure 1) which were the confluence of Batang Ai and Engkari River (A6) which is the inflow; two stations 50 m from existing cage aquaculture operations identified as potential cage culture sites, Teluk Telaus (C14) and Teluk Pudai (D15); and an abandoned and new small aquaculture site (B9) on 28^{th} June 2014 (dry season) and 30^{th} January 2015 (wet season). There were 1000 cages and 216 cages nearby C14 and D15 respectively. As for B9, cages were operating for five years before abandonment in January of 2012; however, during the June 2014 and January 2015 samplings, there were some cages which just started operation.

In-situ water quality parameters such as dissolved oxygen (DO), transparency, electrical conductivity (EC), pH and temperature were measured. DO and temperature were measured using a DO meter and probe (YSI Pro 20) at every meter interval up to 20 m depth. EC was determined using a conductivity meter (EUTECH Instruments), transparency or secchi disc depth (SD) of water was measured by using a secchi disc and pH was determined using a pH meter (WalkLAB TI9000). In addition, depth was measured using a digital sounder (PS-7, HONDEX). Water samples were collected at 0.2 m, 10 m and 20 m where a Wildco 1920-H65 water grab sampler was used at 10 m and 20 m depths. All water samples were stored in HDPE bottles and placed in cooler boxes filled with ice for transportation to the laboratory for analysis. All water samples were analysed in triplicate.

Analysis of five-day biochemical oxygen demand (BOD_5) began in the field. Parameters such as BOD_5, total suspended solids (TSS), soluble reactive phosphorus (SRP), total phosphorus (TP), total sulfide (TS), chemical oxygen demand (COD) and chlorophyll-*a* (Chl-*a*) were determined according to standard methods (APHA 1998). Nitrate-nitrogen (nitrate-N), nitrite-nitrogen (nitrite-N) and total ammonia nitrogen (TAN) were analysed according to Hach (2007). COD was analysed according to the closed reflux method where titration with 0.025 M standard ferrous ammonium sulfate titrant (FAS) was carried out. For TSS, 1 L sample was filtered through a glass-microfibre disc (MGG, Sartorius stedim) and the filter was dried in an oven at 103-105°C until constant weight. For Chl-*a,* filtration was done through a 0.7 µm retention glass-microfibre discs (Sartorius) and spectrophotometric measurement using trichromatic methods was followed whereby the absorbance of the extract was determined using a spectrophotometer (Jasco V630).

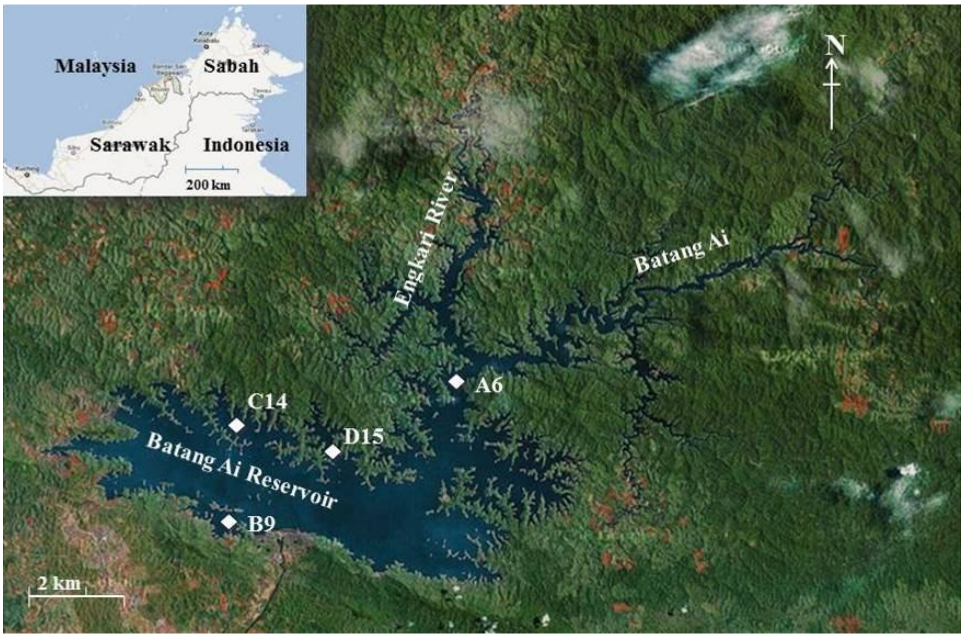

Figure 1. Map of sampling stations at Batang Ai Reservoir (Resource from http://maps.google.com).

Nitrate-N and nitrite-N were analysed according to the cadmium reduction method and diazotization method respectively (Hach 2007). The water sample was distilled before TAN analysis using Nessler's method (Hach 2007). TAN was determined using a spectrophotometer (Jasco V630) at the 425 nm wavelength. SRP was analysed using the ascorbic acid method. For total phosphorus, persulfate digestion method was used to digest the sample followed by ascorbic acid method of phosphate determination. TS was analysed using the methylene blue method. Water quality parameters for each station in Batang Ai River and Reservoir were classified according to National Water Quality Standards for Malaysia (NWQS). Water Quality Index (WQI) was calculated based on six water quality parameters, namely, pH, DO, TSS, BOD, COD and TAN (DOE 2010). The trophic state was determined based on Trophic State Index (TSI) (Devi Prasad & Siddaraju 2012; NALMS 2015) as expressed in Equation [1].

$$TSI = [TSI\ (SD) + TSI\ (Chl\text{-}a) + TSI\ (TP)]\ /\ 3 \qquad [1]$$

The significant difference of each parameter between seasons was conducted by using two-way ANOVA. The comparison of each parameter among stations and within stations for each trip was analysed using one-way ANOVA followed by Tukey's test. Bivariate correlation between the parameters was conducted. SPSS 21.0 was used for all statistical analyses.

Results. The depth of the stations in the reservoir in June and January ranged from 25 to 39 m and 36 to 42 m respectively. Water transparencies of the sampling stations were 3.00-3.63 m in June and 2.31-3.39 m in January (Table 1). In both sampling months, the transparency was the highest at station D15 and the lowest at station C14. Mean transparency in January (2.77 m) was significantly lower compared to June (3.25 m) ($p < 0.0005$).

Table 1

Water transparency (SD) of the stations in June 2014 and January 2015

Station	*Mean SD (m)*	
	June 2014	*January 2015*
A6	3.35±0.00	2.88±0.02[a]
C14	3.00±0.00	2.31±0.01[b]
D15	3.63±0.00	3.39±0.01[c]
B9	3.00±0.00	2.48±0.01[d]

Means in the same column with same alphabets are not significantly different at 5% level; means without alphabets indicate that their replicates are equal in value.

The temperature was higher in value in June (25.5-31.3^oC) than in January (25.6-30.4^oC) and the profiles (Figure 2a, b) were different. The June profile shows that there was a very small decrease from the surface (30.7-31.3^oC) to 8 m depth for stations C14 and B9 while at the other two stations it remained quite constant. This was followed by a steep decrease from 8-10 m followed by a gradual decrease to 25.5^oC at 20 m depth (Figure 2a). However, January temperatures from the surface (29.1-30.4^oC) to 10 m were lower than June and there was more decrease from 1 m to 10 m depth. It was followed by a steeper drop in temperature of about 1^oC followed by a linear decline in temperature to 25.6-25.9^oC at 20 m depth with the slope similar to the top 10 m (Figure 2b). Thus, thermoclines in January (10-12 m) were much less pronounced and deeper than June (8-10 m). In addition, the metalimnion temperature range of June and January were different, namely, 30.4–31.3^oC and 27.7-30.4^oC respectively.

DO profiles show ranges of 0.05-7.52 mg L^{-1} in June and 0.30-6.60 mg L^{-1} in January with decreasing trend as depth increased in both sampling months (Figure 2c, d). However, the profiles were different among stations and between seasons. In June, the top 7-8 m water column DO was quite constant with A6 showing consistently higher value and B9 lower values than the other stations. However, as depth increased further, DO dropped drastically from about 6 mg L^{-1} to as low as 0.05 mg L^{-1}. The difference in DO profile among the stations were, A6 showed the highest DO in particular below 10 m depth and C14 shows shallower oxycline of 7-9 m than the other stations (8-11 m). In January, the profiles were different from June in that DO values at the top 7-8 m water column (5.1-5.9 mg L^{-1}) were lower than those in June, the decline in DO was more gradual and the DO below the thermocline (0.3-1.5 mg L^{-1}) was higher than June (< 0.1 mg L^{-1}). In January, at 10 m depth, DO was 2.5-4.3 mg L^{-1} which was higher than the corresponding June DO (0.05-3.0 mg L^{-1}). Therefore, the January oxycline (8-11 m) was deeper than June at all stations except A6 (7-9 m). At A6, below the oxycline, DO value was higher in January than June. The DO readings showed a significant difference between seasons ($p = 0.005$).

pH values ranged from 6.21 to 7.46 in June and from 5.92 to 7.27 in January. It was in a decreasing trend as depth of water increased at all stations in both sampling months, except for station D15 which showed a different trend in January whereby pH decreased from 0.2 m to 10 m followed by an increase to 20 m (Figure 3a, b). Comparing among stations, D15 recorded the lowest pH at the subsurface (6.85) and 10 m depth (6.45) in June and in all the three depths in January. Besides that, C14 showed the lowest pH of 6.21 at 20 m depth among the stations in June (Figure 3b). pH of stations A6, C14 and B9 at 20 m depth showed significantly lower values than 0.2 m and 10 m depths ($p < 0.0005$) in June. At 0.2 m and 10 m of station D15 significantly lower pH than other stations in June and January ($p < 0.0005$) was observed. In January, 0.2 m pH of A6, C14 and D15 was significantly higher than 10 m ($p < 0.0005$). Mean pH were significantly different between seasons ($p = 0.039$).

June EC ranged from 4.71 to 75.50 µS cm^{-1} whereas January showed a smaller range (20.50 to 67.10 µS cm^{-1}). In June, among the stations, EC at C14 was the highest at 10 m (75.5 µS cm^{-1}) and 20 m (67.10 µS cm^{-1}) (Figure 3c) and among depths, EC was the highest at 20 m of all the stations. In January, EC showed an increasing trend at all stations as depth increased and the highest were recorded at 20 m depth (Figure 3d). In both trips, at 20 m depth, C14 showed the highest EC and A6 the lowest EC. All stations except C14 showed significantly higher EC at 20 m compared with 0.2 m and 10 m depths in both seasons ($p < 0.0005$).

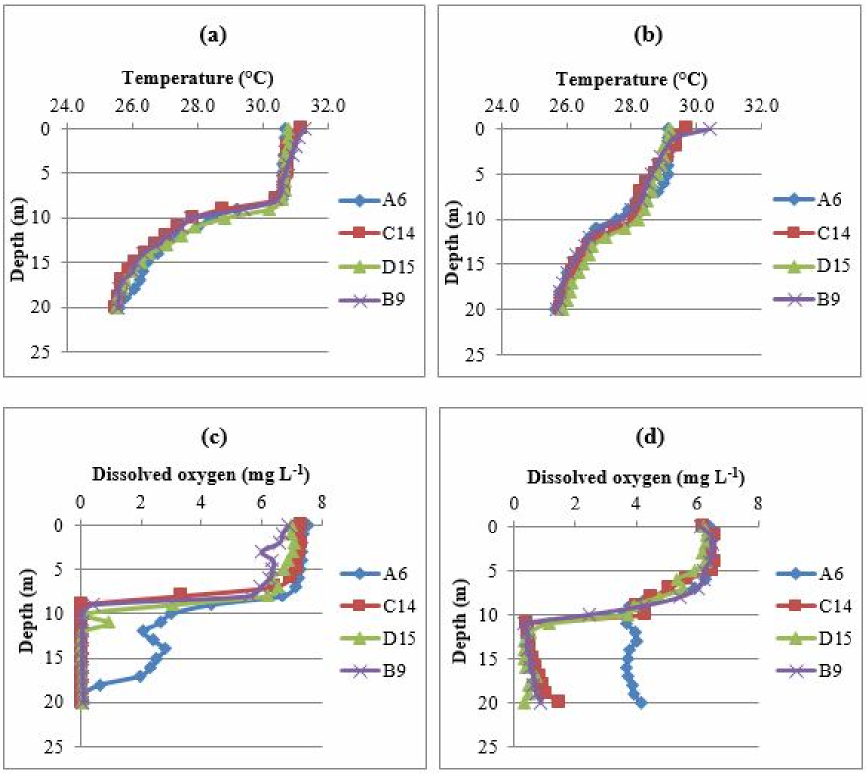

Figure 2. Temperature and dissolved oxygen profiles at four stations in (a), (c) June 2014 and (b), (d) January 2015.

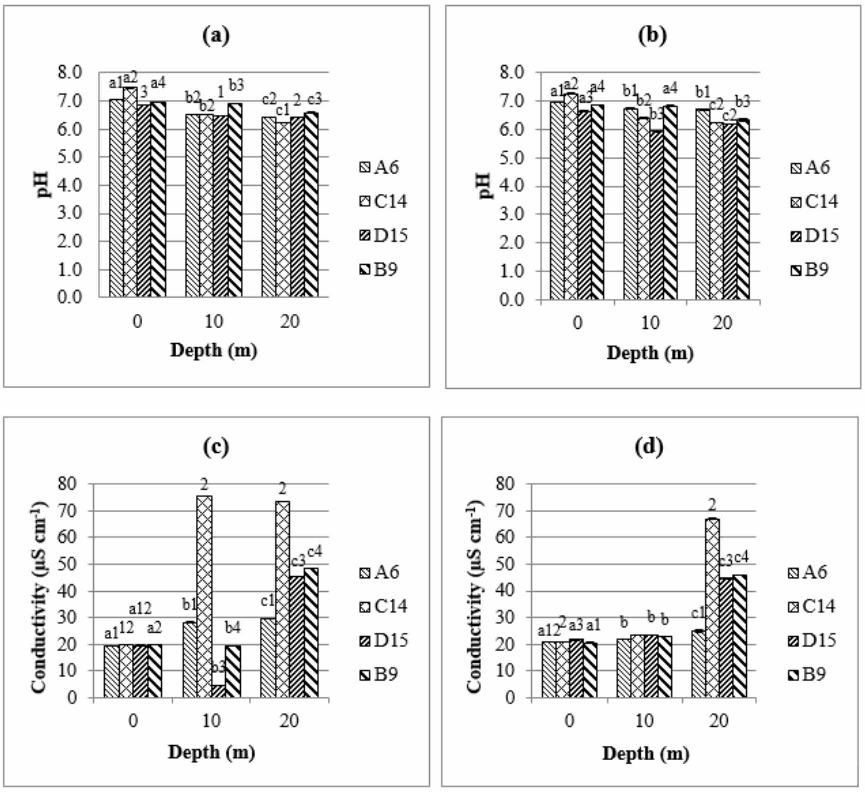

Figure 3. Mean pH and specific conductivity at 0.2 m, 10 m and 20 m depth of the four stations in (a), (c) June 2014 and (b), (d) January 2015. Means from the same station with same alphabets and means from the same depth with the same number were not significantly different at 5% level. Bars without alphabet or number indicate that their replicates were equal in value.

June and January BOD_5 were similar in range, 5.00-9.53 mg L^{-1} and 5.00-9.40 mg L^{-1} respectively. A6 and C14 showed significantly higher BOD_5 at 10 and 20 m depths than 0.2 m depth ($p < 0.05$) (Figure 4a). In addition, station B9 at 0.2 m depth recorded significantly higher BOD_5 than the other stations ($p < 0.05$), but there was no significant difference among the depths ($p = 0.198$). In January, the highest two BOD_5 values were recorded at station D15 at 0.2 m and 10 m depths, respectively. At 0.2 m depth, stations A6, C14 and D15 showed significantly higher BOD_5 than station B9 (Figure 4b). At 10 m depth, C14, D15 and B9 showed significantly higher BOD_5 than A6 while at 20 m depth, C14 and D15 showed significantly higher BOD_5 than the other two stations.

June COD range of 66.6-88.8 mg L^{-1} was higher than January (37.3-69.3 mg L^{-1}) (Figure 4c, d). C14 showed the highest at all depths (Figure 4c). The highest COD at 10 m and 20 m was A6 and at that station, there was no significant difference among the three depths (Figure 4d). At 0.2 m depth, COD at stations A6 and C14 were significantly higher than D15 (Figure 4d). Between the two seasons, there was a significant difference in COD ($p < 0.0005$).

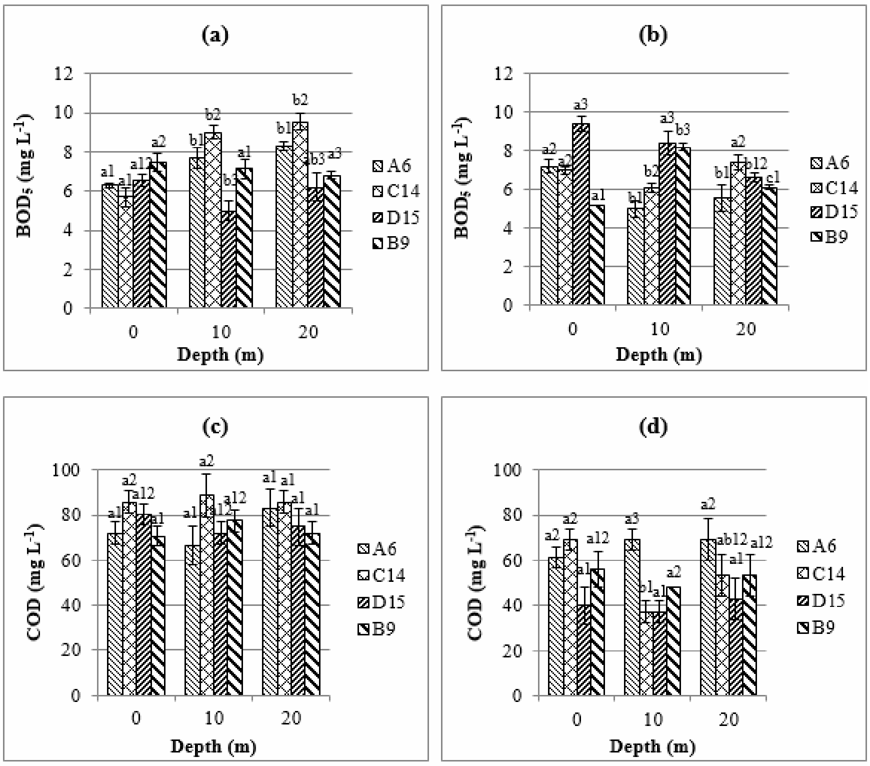

Figure 4. Mean five-day biochemical oxygen demand (BOD_5) and chemical oxygen demand (COD) at 0.2 m, 10 m and 20 m depths of the four stations in (a), (c) June 2014 and (b), (d) January 2015. Bars from the same station with same alphabets and bars from the same depth with the same number were not significantly different at 5% level.

TSS increased with depth at most of the stations (Figure 5a, b). June TSS, 1.3-16.3 mg L^{-1}, was lower than January values (3.0-18.3 mg L^{-1}). In June, the highest TSS occurred at station C14 at 20 m depth and the second highest was at station B9 at 10 m depth. TSS at 20 m depth of all stations were significantly higher than 0.2 m depth except station D15 which shows no significant difference among the depths. In January, A6, C14 and B9 TSS increased as the water gets deeper (Figure 5b). C14 showed the highest TSS at both 10 m and 20 m. At 20 m depth, TSS at all stations were significantly higher than 0.2 m except station D15 which shows no significant difference. TSS difference between the seasons was not significant ($p = 0.093$).

In both seasons, mean Chl-*a* was the highest either at 10 m or 20 m depth for all stations except A6 (Figure 5c, d). In June, Chl-*a* range was 0.22-24.11 mg m^{-3} (Figure 5c). At station C14, Chl-*a* at 10 m and 20 m were 30 and 3 times that of 0.2 m respectively. In addition, D15 Chl-*a* at 10 m and 20 m were 15 times that of 0.2 m and at B9, as we move deeper to 10 m and 20 m, the increase in Chl-*a* were 5 and 10 times respectively. In general, there was a decrease in Chl-*a* from June to January (0.15-12.94 mg m^{-3}) with a couple of exceptions, the most noticeable being 10 m depth at station B9 and 20 m depth at station C14. The means were significantly different between seasons ($p = 0.008$).

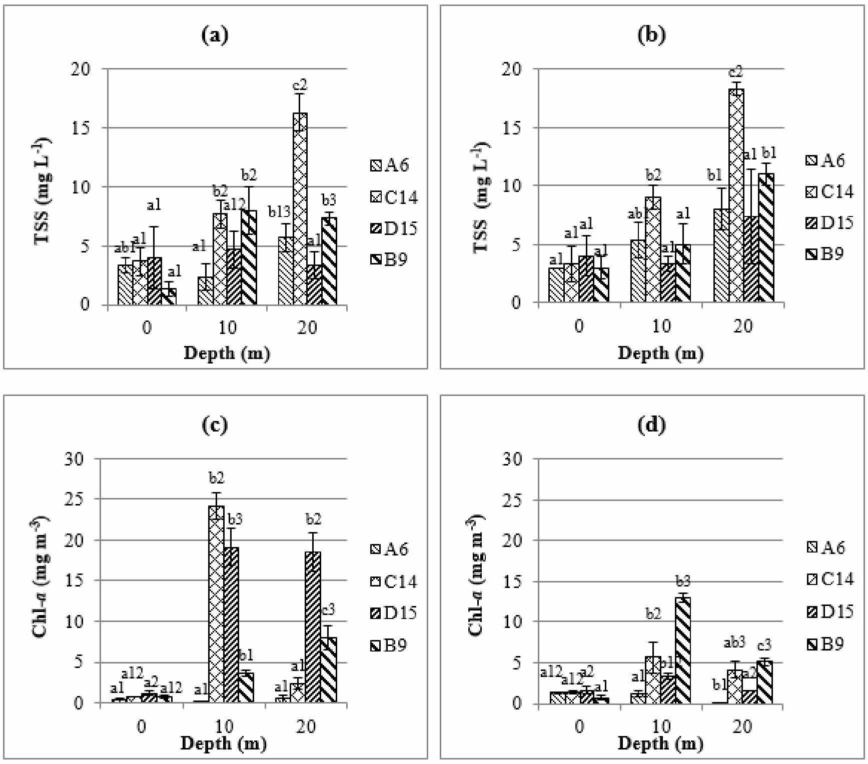

Figure 5. Mean total suspended solids (TSS) and chlorophyll-*a* (Chl-*a*) at 0.2 m, 10 m and 20 m depths of the four stations in (a), (c) June 2014 and (b), (d) January 2015. Means from the same station with same alphabets and means from the same depth with the same number were not significantly different at 5% level.

Nitrate-N of June and January were 0.017-0.240 mg L^{-1} and 0.043-0.109 mg L^{-1} respectively (Figure 6a, b). In June, C14 showed the highest concentrations of 0.240 mg L^{-1} at 0.2 m and it decreased significantly to 0.064 mg L^{-1} and 0.017 mg L^{-1} at 10 m and 20 m depths respectively. On the other hand, at station A6, the concentration increased significantly from 0.2 m to 20 m depth in both seasons. Nitrite-N in June (Figure 6c) was significantly higher ($p = 0.001$) than January mean (Figure 6d). Among the stations studied, in June, at 10 m depth, C14 mean was significantly higher than A6 and D15 but not significantly different from B9; at 20 m depth, D15 mean was significantly higher than all the others. In January, at 20 m depth, A6 mean was significantly lower than all the other stations ($p < 0.05$).

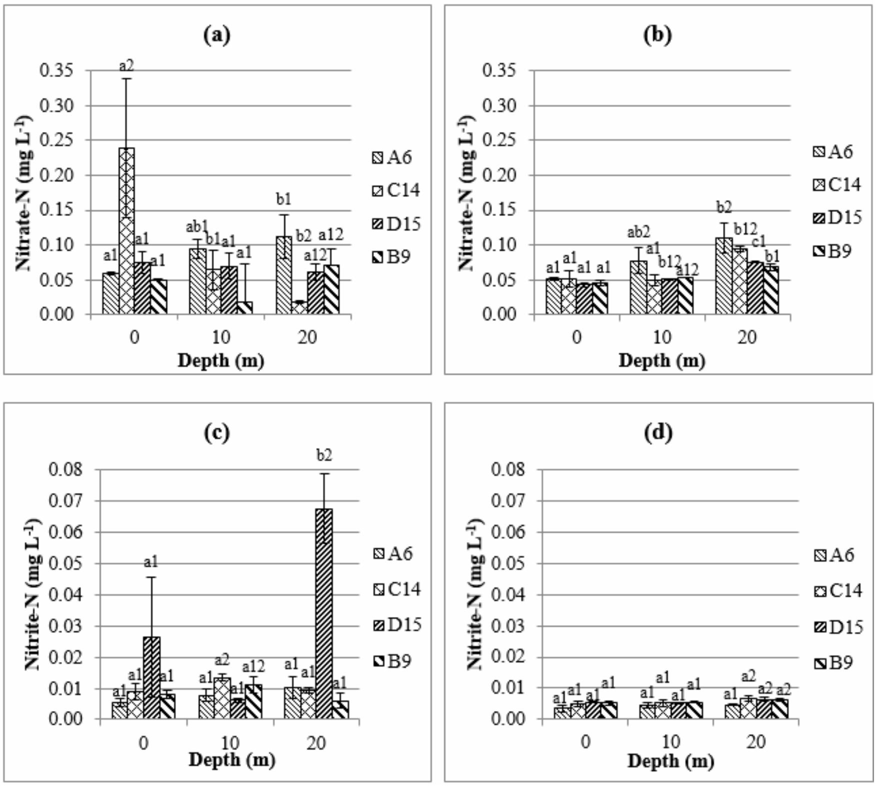

Figure 6. Mean nitrate-nitrogen (nitrate-N) and nitrite-nitrogen (nitrite-N) at 0.2 m, 10 m and 20 m depths of the four stations in (a), (c) June 2014 and (b), (d) January 2015. Means from the same station with same alphabets and means from the same depth with the same number were not significantly different at 5% level.

June TAN of 0.094-1.942 mg L^{-1} (Figure 7a) was much higher than January values (0.003-1.233 mg L^{-1}) (Figure 7b) and the difference was significant ($p < 0.0005$). In June, C4 showed the highest TAN at all depths and the concentration increased significantly with depth ($p < 0.05$). Other than this station, B9 also showed significant increase with depth and for D15, the concentration ranked in increasing order of 0.2 m < 20 m < 10 m and the differences between the means were significant ($p < 0.05$). June A6 TAN means were the lowest among the stations and there was no significant difference among the three depths ($p > 0.05$). In January (Figure 7b), the highest and the second highest TAN were recorded at 20 m depth of C14 and D15 respectively, whereas the lowest TAN was also observed at 0.2 m depth of A6. In addition, all the stations showed significantly higher TAN at 20 m depth than 0.2 m depth ($p < 0.05$).

TS in June (Figure 7c) was the lowest in concentration at 0.2 m depth and means at three stations, A6, C14, and D15 increased to the highest level at 10 m depth whereas B9 showed the highest at 20 m. The means in June ranged from 0.2 to 77.4 µg L^{-1} and C14 showed significantly higher TS than the other stations at 0.2 m and 20 m depths. In addition, TS at 10 m depth of D15 was significantly higher than A6 and B9 ($p < 0.05$) but not different from C14 ($p > 0.05$). At 20 m depth all three stations, C14, D15 and B9 show significantly higher means than A6. January mean TS (Figure 7d) were more even out among stations and among depths when compared with Jun with narrower range of 6.7-30.3 µg L^{-1}. Thus, January means show a noticeable increase in 0.2 m TS at A6, D15 and B9; 10 m depth of B9 and 20 m depth of A6, from June means. Though more uniform concentrations, 20 m depth still showed significantly higher concentrations than 0.2 and 10 m depths at all stations except A6 which shows no significant difference between 10 and 20 m depths. Mean TS values were significantly different between seasons ($p = 0.001$).

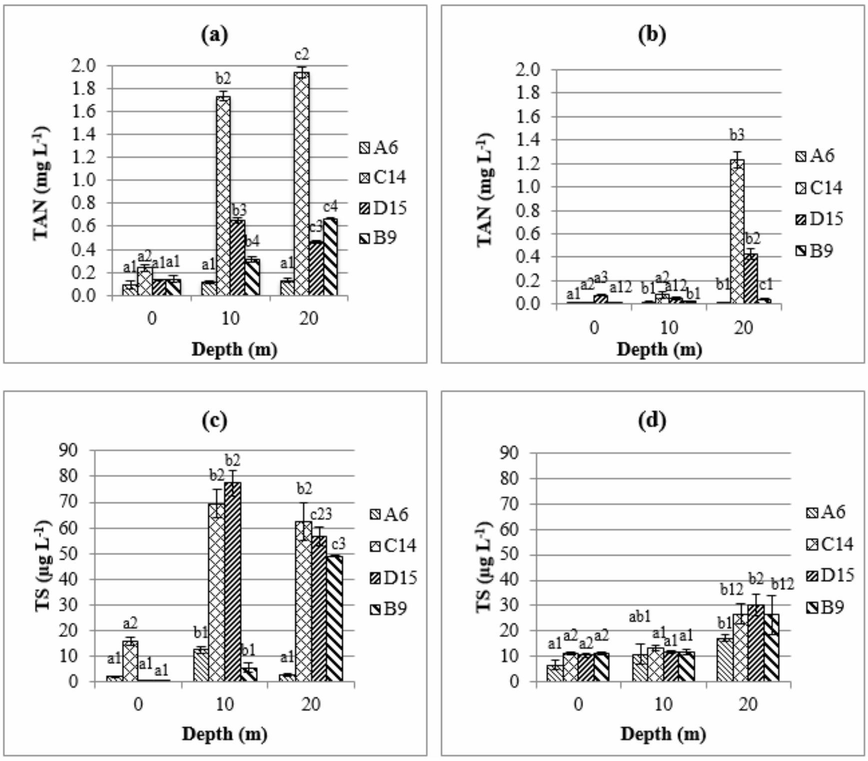

Figure 7. Mean total ammonia nitrogen (TAN) and total sulphide (TS) at 0.2 m, 10 m and 20 m depths of the four stations in (a), (c) June 2014 and (b), (d) January 2015. Means from the same station with same alphabets and means from the same depth with the same numbers were not significantly different at 5% level.

In June, mean SRP ranged from 1.7 to 28.8 µg L^{-1} and the highest SRP was at 0.2 m of B9, while the lowest SRP was at 10 m of D15 (Figure 8a). At station A6, SRP increased significantly with depth to 20 m where its mean ranked the second highest (18.1 µg L^{-1}). At D15, the highest was also at 20 m. C14 means (2.7-3.6 µg L^{-1}) were low and not significantly different among depths ($p > 0.05$). The highest January SRP mean (25.6 µg L^{-1}) was observed at 0.2 m of D15 whereas the other means were much lower ranging from 1.1 to 6.2 µg L^{-1} (Figure 8b). In addition, all the stations showed significantly higher SRP at 0.2 m depth except B9 ($p < 0.05$). Mean SRP values showed a significant difference between seasons ($p = 0.015$).

TP in June (Figure 8c) ranged from 7.9 to 151.4 µg L^{-1} and most of the means were higher than January means (2.3-93.9 µg L^{-1}) (Figure 8d). In June, C14 showed the highest TP at all depths and the differences with the other stations were significant at all depths ($p < 0.05$). The lowest June TP was recorded at B9 at 10 m depth. In addition, TP of A6, C14 and D15 showed the highest mean at 10 m. The highest TP in January was observed at D15 at depth of 20 m followed by 0.2 m of A6 (87.32 µg L^{-1}) and 20 m of C14 (78.5 µg L^{-1}). As we go deeper, TP increases significantly with depth at three stations, C14, D15 and B9 whereas the opposite occurred at A6. Mean TP were significantly different between seasons ($p = 0.015$).

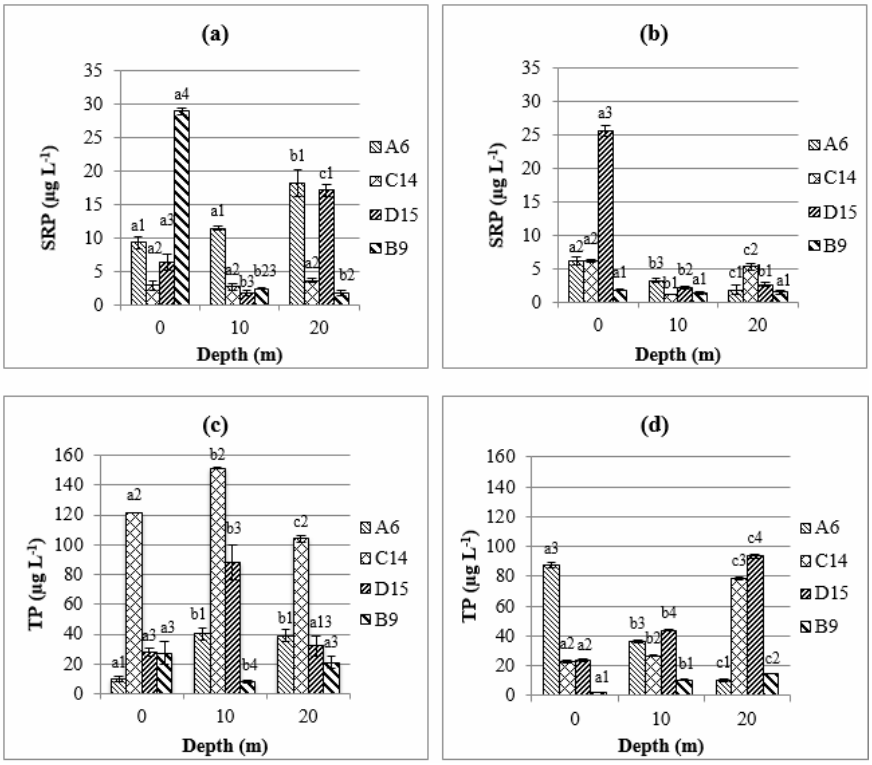

Figure 8. Mean soluble reactive phosphorus (SRP) and total phosphorus (TP) at 0.2 m, 10 m and 20 m depths of the four stations in (a), (c) June 2014 and (b), (d) January 2015. Means from the same station with same alphabets and means from the same depth with the same numbers were not significantly different at 5% level.

Discussion. Water depths measured at the stations and visual observations of the banks of the reservoir show that the water level was higher in January than June due to the rainy season. Rainy season also affected the transparency of all stations in January where significantly lower readings than June were observed as particles came from eroded soil around the reservoir other than the inflow. Most of the water quality parameters showed lower and more uniform January concentrations as a result of dilution and mixing with a few exceptions such as 0.2 m and 10 m BOD_5, 0.2 m SRP and 20 m TSS at C15. Those exceptions indicate that dilution is not able to offset the input. Total phosphorus at A16 in rainy season was an exception too in terms of concentration and trend; it decreased with depth which is the opposite of the other stations in the reservoir due to the surface runoff input and dilution at the bottom which is supported by evidence of higher bottom DO from rivers inflow. Mixing of rain water, rivers inflow and runoff and wind effects in rainy season have also resulted in more uniform concentrations of nitrate, nitrite and total sulphide among stations at all depths.

In both seasons, nitrate-nitrogen shows the highest concentrations at 10 m and 20 m depths at station A6 indicating input from rivers especially Engkari River where there are settlements in longhouses. At A6, other than nitrate, higher BOD_5 than stations D15 and B9 at 10 m and 20 m depths, higher 0.2 m BOD_5 than station C14 in June, higher 20 m COD in June and the highest COD at depths of 10 m and 20 m were observed. A6 SRP was significantly higher at 10 m and 20 m depth than B9 and C14 during dry season. All these are due to contributions of nitrate, phosphate and organic matter from partially treated household grey water as studies of household wastewater in Kuching showed high nitrate, phosphate and BOD_5 concentrations (Ling et al 2010, 2012a).

In June, thermocline lies between 8-10 m except at C14 which was shallower (5-7 m), and the temperature of epilimnion was higher, above 30^oC because it was the dry season and rain was rare for four months before the sampling except the night before sampling when it rained. The confluence of Batang Ai River and Engkari River showed lower water temperature as it receives the inflow water from Batang Ai and its tributaries. In January, the temperature profile shows a different trend mainly at the top 10 m depth where temperature is lower and there is an obvious negative gradient in temperature with depth. Due to the rainy season, there is more cloud cover, thus, less heat transfer to the water from solar radiation which leads to lower temperature. In addition, input of rainwater and runoff from the watershed is cooler. As a result, the temperature decrease rate in the top 10 m is constant which is almost similar to hypolimnion decrease rate. Thus, a deeper and less distinct thermocline was observed.

DO for all stations decreased with depth as there is atmospheric diffusion at the epilimnion but little aeration occurred as depth increases. DO profiles show that DO value deeper than 7 m could drop below 5 mg L^{-1} which is not suitable for healthy fish growth (Lawson 1995). A6 showed higher dry season DO and wet season bottom DO than three other stations which is attributed to inflow from upstream and Engkari River where the inflow water along the bottom of upstream forms an interflow in the hypolimnion (Hayami et al 2008). With the depth of fish cage 3-7 m deep (Effendie et al 2005), should there be any overturn of water, fish in the cages may not have enough oxygen to survive as occurred in Cirata Reservoir where a strong vertical mixing induced by wind caused massive fish kill because of the shallowest oxycline (5-7 m) (Hayami et al 2008). DO observed in the present study (0.05-7.52 mg L^{-1}) is lower but close to the range of 0.26 to 8.45 mg L^{-1} reported previously (Nyanti et al 2012).

Even though C14 and D15 were not aquaculture sites, they showed signs of impacts from existing cage culture operations as evidenced by C14 showing the lowest transparency, the highest 20 m conductivity, BOD_5, TSS and TAN, the lowest 20 m pH and the highest subsurface pH in both seasons. In addition, in the dry season of June it showed the highest 10 m conductivity, BOD_5, COD, Chl-*a*, and TAN and the highest subsurface COD, nitrate-N, TAN, total sulphide. Another notable observation at C14 was the highest TP at all depths during dry season and the concentrations at the C14 depths were 2.5-18.8 times that of station B9, the abandoned cage culture site at corresponding depths. In addition, at D15, the concentrations at the depths were 1.6-11 times that of station B9 except 0.2 m depth during dry season. Other than that, pH decreased as depth increased at all stations with more decrease for C14 and D15 than other stations. C14 showed the shallowest oxycline of 7-9 m whereas the others were from 8-10 m. All these observations point to the impacts of the input of cage aquaculture related waste such as fish physiological waste, uneaten food, and dead fish. In addition, there are also contributions from stomach waste and offal from fishes de-gutted on-site prior to marketing them. Suspended solids resulting from those waste decreased transparency, increased total suspended solids and nutrients such as total phosphorus and ammonia-nitrogen. As the solids moved away and downward it provided nutrients for phytoplankton in the mixing zone of the water column and as decomposition occurred at the deeper regions, conductivity increased and pH was lowered. This is supported by significant correlation between conductivity and TSS in June ($p = 0.016$, $r = 0.647$, $N = 12$) and January ($p < 0.0005$, $r = 0.895$, $N = 12$). As it was anoxic at the deeper region at 10 m and 20 m depth during dry season, nitrification was limited and accumulation of ammonia occurred as observed. Studies have shown that cultured fish only assimilate about 14.8% of TN and 11.0% of TP, and loading rates of TN and TP to the environment were 0.12-0.16 kg and 0.025-0.035 kg per kg of fresh fish produced respectively (Guo et al 2009) and nutrient loss from tilapia fish cages in Lake Malawi ranged from 59 to 80% for nitrogen and from 85 to 92% for phosphorus (Gondwe et al 2011). These explain the high nitrogen and phosphorus at C14 and D15. Ling et al (2013b) also reported that TP was higher in dry season than wet season and the organic particulate form was the highest near another aquaculture site in the reservoir. In this study, the highest concentration of Chl-*a* was found at 10 m depth in the thermocline region instead of subsurface due to photoinhibition in the mixed layer where carbon fixation is suppressed (Long et al 1997). Similar observation of deep Chl-*a* was also observed previously in this reservoir (Ling et al 2013a) and in Demirdoven Dam Reservoir, Turkey (Kivrak & Hasan 2005). For nitrate-nitrogen, 46% of the current concentrations exceeded the maximum (0.06 mg L^{-1}) reported previously (Nyanti et al 2012) but for nitrite-N, they are in the range of 0.001 to 0.053 mg L^{-1}.

Decomposition of organic matter in anoxic condition by sulphur reducing bacteria results in the formation of hydrogen sulphide (Baxter 1977) which explains the observations of higher sulphide at 10 m and 20 m depths except A6. This is supported by the significant negative correlation of DO and sulphide in June ($p = 0.007$, $r = -0.728$, $N = 12$) and January ($p = 0.001$, $r = -0.812$, $N = 12$). At A6, in dry season, the sulphide at 20 m depth was significantly lower than 10 m depth due to inflow. For stations in the reservoir, in dry season, there were pockets of high sulphide observed at 10 m depth of C14 and D15 where the concentrations were 9-20 times higher than A6 and B9 at the same depth and also at 20 m depths, the concentrations were higher most likely due to the contributions of fish waste near to those sites and less exchange of water due partial enclosure by land mass. On the other hand, such high concentrations at 10 m depth and 20 m were not observed in rainy season, however, 0.2 m hydrogen sulphide at all the stations increased in wet season most likely due to mixing. Compared with the previous report, 21% of total sulphide readings in the present study (0.15-77 μg L^{-1}) exceeded the maximum of the previous range (0.33-32 μg L^{-1}) (Ling et al 2012b) and they all occurred during dry season at deeper regions but not A6, the inflow. Un-ionized hydrogen sulphide in the present study is 0.07-59.4 μg L^{-1} indicating its potential hazard on fishes in the reservoir as un-ionized hydrogen sulphide is toxic to fish at different life stages even at low concentration, for example, 10 μg L^{-1} can inhibit reproduction (Lawson 1995; Bagarinao 1992). Compared with USEPA (1986) which recommended limit of 2 μg L^{-1}, all the values exceeded that limit except the 0.2 m dry season hydrogen sulphide at A6, D15 and B9 (0.07-0.95 μg L^{-1}). Even though Bagarinao & Lantin-Olaguer (1998) reported that *Oreochromis mossambicus* is rather tolerant to hydrogen sulphide, the combined effects of low DO and high sulphide could be lethal to *Oreochromis* sp. cultured in cages of Batang Ai if there is any overturn.

Compared to other studies, the dry season transparency values are in the range reported for West Malaysian Reservoirs (0.3-3.8 m) and between that of Sg Terip Lake (2.7 m) and Kenyir Lake (3.8 m) (Sharip et al 2014). Ninety six percent (96%) of total phosphorus (2-151 μg L^{-1}) in the present study exceeded the range of 5-20 μgP L^{-1} for most natural surface water (Chapman 1996) and all depths TP at C14 exceeded the maximum 0.2 m TP (67 μg L^{-1}) observed at another aquaculture station in 2011-2012 (Ling et al 2013b) attributable to the larger size of operation and on-site fish processing in the present study. Dry season subsurface TP (10-121 μg L^{-1}) in the present study is lower than those (260-3430 μg L^{-1}) reported by Sharip et al (2014) for lakes in West Malaysia. The range of TSS in the present study (1.3-18.3 mg L^{-1}) is close to the TSS reported previously (1.3-11.0 mg L^{-1}) (Ling et al 2013a).

A number of parameters exceeded the Class II limits of NWQS meant for the protection of sensitive aquatic organisms (DOE 2010). They are, pH of station D15 classified as Class III (pH 5-9); BOD for all stations exceeded the limit of Class II (3 mg L^{-1}) and thus fall in Class III (3-6 mg L^{-1}) or Class IV (6-12 mg L^{-1}); COD exceeded Class II limit of 25 mg L^{-1} with most of the stations in Class IV (50-100 mg L^{-1}); June TAN at 10 m and 20 m depths at stations D15 and B9 were classified as Class III (0.3-0.9 mg L^{-1}) and C14 was classified as Class IV (0.9-2.7 mg L^{-1}) whereas January TAN of station D15 and C14 at 20 m depth fall into Class III (0.3-0.9 mg L^{-1}) and Class IV (0.9-2.7 mg L^{-1}) respectively. WQI of stations (Table 2) indicates that the reservoir water is more polluted during dry season than wet season and it is important to note that C14 and D15 showed worse water quality than A6 and B9.

When the water quality of 0.2 m is classified based on TSI (Devi Prasad & Siddaraju 2012; NALMS 2015), the stations were found to be oligotrophic or mesotrophic (Table 3). The value of TSI of A6 in June and station B9 in January were in the range of 30-40 which is classified as oligotrophic. The other stations were classified as mesotrophic with TSI range of 40-50 in both sampling months. However, TSI (TP) of A6 in rainy season and C14 in dry season were at eutrophic and hypereutropic states.

Table 2

Water Quality Index (WQI) of all stations in June 2014 and January 2015

Station	*Depth (m)*	*June 2014*			*January 2015*		
		WQI	*Class*	*Status*	*WQI*	*Class*	*Status*
A6	0.2	78.3	II	Slightly polluted	81.5	II	Clean
	10	60.0	III	Slightly polluted	75.4	III	Slightly polluted
	20	51.5	III	Polluted	74.2	III	Slightly polluted
C14	0.2	74.9	II	Slightly polluted	80.5	II	Clean
	10	40.3	IV	Polluted	72.4	III	Slightly polluted
	20	38.3	IV	Polluted	51.7	IV	Polluted
D15	0.2	76.1	II	Slightly polluted	80.7	II	Clean
	10	53.3	III	Polluted	62.7	III	Slightly polluted
	20	51.1	III	Polluted	55.9	III	Polluted
B9	0.2	75.8	II	Slightly polluted	84.0	II	Clean
	10	70.8	III	Slightly polluted	74.0	III	Slightly polluted
	20	48.9	III	Polluted	60.2	III	Slightly polluted

Table 3

Trophic State Index (TSI) of the stations in June 2014 and January 2015

St	*TSI (SD)*	*TSI (Chl-a)*	*TSI (TP)*	*TSI*	*Attribute*	*TSI (SD)*	*TSI (Chl-a)*	*TSI (TP)*	*TSI*	*Attribute*
	June 2014					*January 2015*				
A6	42.6	19.6	37.0	33.1	Oligotrophy	44.8	33.6	68.6	49.0	Mesotrophy
C14	44.2	28.1	73.3	48.6	Mesotrophy	47.9	33.1	49.4	43.5	Mesotrophy
D15	41.4	32.4	52.0	42.0	Mesotrophy	42.4	35.2	49.9	42.5	Mesotrophy
B9	44.2	28.1	51.9	41.4	Mesotrophy	46.9	27.3	16.2	30.1	Oligotrophy

Conclusions. The present study shows that dry season water quality is worse than rainy season as subsurface water is classified as clean during wet season but slightly polluted during dry season. At 10 m of two stations near aquaculture sites, C14 and D15, dry season water is polluted attributable to aquaculture waste nearby and the lack of mixing. Though DO values at the top 7 m column met the minimum of 5 mg L^{-1} required for sensitive aquatic organism, pockets of high sulphide were observed. This could affect fish survival and growth. Besides that, the trophic state at C14 and D15 were mesotrophic regardless of season. It is recommended that aquaculture sites should be properly spaced and fish processing in the reservoir should be avoided.

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