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Effects of Intermittent Artificial Circulation in Summer Months on Chlorophyll *a* Concentration in a Small Eutrophic Impoundment

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Abstract: Impact of artificial circulation on total algal biomass (as chlorophyll *a* integrated over depth) in a small eutrophic impoundment in the Northern Great Plains, was monitored over three consecutive summers. Chlorophyll *a* concentration and nutrient concentrations were measured at depths along water columns under aerated and non-aerated conditions at four sampling locations. Aeration destratified the water column, which resulted in homogenous chlorophyll *a* concentration throughout the water column and across the four sampling locations. The non-aeration resulted in concentration gradient along the water column at all sampling locations. Overall, the artificial circulation resulted in an overall reduction of chlorophyll *a* concentration along the water column and across sampling locations. Reduced chlorophyll *a* might be attributed to a shift in dominant phytoplankton species. For example, increased mixing by aeration may have favored dinoflagellates or green algae over cyanobacteria. In conclusion, the artificial circulation in this impoundment was effective in reducing total algal biomass.

Keywords: Chlorophyll *a*, Eutrophication, Artificial Circulation, Nutrients, Algal Bloom

Introduction

This paper analyzes the effects of intermittent artificial circulation in summer months on chlorophyll *a* concentration in a small eutrophic impoundment in North Dakota, USA. Algal blooms are frequently caused by eutrophication due to enrichment of nutrients, mainly nitrogen (N) and phosphorous (P). These blooms contribute to oxygen depletion in the hypolimnion. Artificial circulation (artificial destratification) has been widely adopted as a restoration method to control algal blooms, especially harmful algal blooms, which produce toxins in eutrophic lakes (Schladow and Fisher, 1995; Berman and Steinman, 1998). Artificial circulation prevents or disrupts thermal stratification in a lake or reservoir and induces destratification.

Artificial circulation affects phytoplankton communities in different ways. For instance, low-light adapted species can become dominant as mixing makes the water more turbid (Burgi and Stadelmann, 2002). Conversely, species susceptible to sedimentation losses such as *Microsystis* sp., will decline in numbers (Lilndenschmidt, 1999; Jungo *et al*., 2001). Therefore, it is possible to alter the phytoplankton population by artificial circulation. The response, however, of total algal biomass to artificial circulation has been quite variable. Pastrok *et al*. (1982) reviewed 40 cases of complete artificial circulation and found that the effect on chlorophyll *a* varied among lakes.

Some studies show that aeration decreases algal biomass through replacement of cyanobacteria, which prefer calm conditions, with dinoflagellates or green algae (Steinberg, 1983; Cooke *et al*., 1993; Hyenstrand *et al*., 1998; Simmons, 1998; Jungo *et al*., 2001; Wetzel, 2001). In other studies, however, aeration resulted in increased algal biomass (Becker *et al*., 2006), due to supply of hypolimnetic nutrients to the epilimnion and expansion of favorable habitat for phytoplankton. In addition, some studies have shown no change in the phytoplankton population after aeration (Barbiero *et al*., 1996; Sherman *et al*., 2000). There is no solid conclusion about whether artificial aeration increases or decreases total algal biomass. A better understanding of the effects of artificial aeration on total algal biomass in the water column is crucial for effective lake management.

This study was carried out in Heinrich- Martin Dam Impoundment (HMDI), located in north central LaMoure County, North Dakota. The HMDI is an important impoundment for bluegill fishery across the state since HMDI is used to maintain blue gill brood stock. An artificial circulation system was installed and operated by the North Dakota Game and Fish Department (NDG&F) in an attempt to increase the Dissolved Oxygen (DO) in the hypolimnion. In a previous study, Overmoe (2008) observed increased weed and algal growth with aeration; however, no systematic study has been done to show whether artificial circulation improves the water quality in HMDI in terms of reduction of total algal biomass. In addition, very few studies in general and no studies of HMDI have monitored chlorophyll *a* variation along depths from surface to bottom and across the sampling locations, including deep and shallow locations and closer to the aeration system as well as away from the aeration system.

Chlorophyll *a* concentration has been used world-wide as a measure of total algal biomass. Algal biomass is one of the widely used variables in limnology to determine the trophic status of lakes (Kasprzak *et al*., 2008; Boyce *et al*., 2010; Ramaraj *et al*., 2013). Algae can be harvested to improve the trophic state of lakes. Algal biomass is used as a raw material for producing food products, biofuels and fertilizers (Hannon *et al*., 2010; Thiamdao *et al*., 2012). Moreover, biomass can be used as a potential alternative to fossil fuels due to its less severe impact on the environment and their ability to renew (Hannon *et al*., 2010; Sgroi *et al*., 2015). The economic value of the lakes is believed to be high and beneficiaries across the United States are willing to pay for the lake water quality (Boyle and Bouchard, 2003; Dziuk and Heiskary, 2003). For instance; in Minnesota, the price of lakefront property has lowered by up to $500 per frontage foot due to declines in water clarity (Dziuk and Heiskary, 2003). Trapani *et al*. (2014) studied the suitability of payment option for the environmental services for the sustainable development of forest territory and reported the possibility of improved forest management options in their study area. Similar approach could be applied for lake water quality improvement as well. However, it is important to manage, preserve and restore the natural environments for human well-being and environmental sustainability. Economics of biomass production and removal and other lake water improvement options such as artificial circulation etc., should be considered only with due consideration for environmental sustainability and human well- being. Environmental assets such as good lake water quality, forests, etc., provide many services for maintaining human well-being. For instance, forest provides a multitude of ecosystem services such as wildlife habitat, preserve biodiversity, water retention, timber and erosion prevention (Fedrowitz *et al*., 2014). Brandt *et al*. (2014) analyzed linkages and tradeoff between ecosystem services and biodiversity in temperate rain forests in the Pacific Northwest of the United States and showed positive linkages leading to species richness. They grouped ecosystems into four categories provision, supporting, regulating and cultural. Landscape aesthetics was one of the ecosystem services they considered which included lakes and rivers among other natural elements. Natural elements largely undisturbed by anthropogenic impact are generally accepted as a great benefit that ecosystem services may offer. Everard and McInnes (2013) call for a systemic solutions approach to water and environmental management. They emphasize the need for low-input, optimized ecosystem service output solutions. According to them, systemic solutions are low-input technologies using natural processes to optimize ecosystem services and their benefits ensuring sustainability.

Artificial aeration is a management option to improve water quality. To evaluate the benefits of artificial circulation technology, a comprehensive study was carried in HMDI under aerated and non-aerated conditions by measuring chlorophyll *a* concentration, which is an indirect measure of total algal biomass, to see whether the existing aeration system suppresses total algal biomass in the water column along with other water quality parameters. This is the first comprehensive study to show the trend of chlorophyll *a* concentration throughout the impoundment under aerated and non-aerated conditions. This information will be crucial for lake managers to provide recommendations and/or alternative methods to the NDG&F to control the eutrophication, maintain fish population and improve the existing aeration system in HMDI.

Materials and Methods

Study Area

The HMDI is a small reservoir with a surface area of 18.8 acres (0.08 km^2) and it is used for recreational activities such as boating and fishing (Wax *et al*., 2008). The average and maximum depths of water are 4.3 and 10 m, respectively (Wax *et al*., 2008).

Aeration System

The HMDI aeration system is an artificial destratification system placed in the deepest area (10 m) of the reservoir (Overmoe, 2008). The HMDI system was installed in 2006 by NDG&F, for the purpose of breaking down the thermal stratification, increasing the DO concentration in the impoundment and improving fish growth (Overmoe, 2008). The aeration system has been in operation since 2006 (with the exceptions of 2009 and 2011) for 5-6 months per year in early summer prior to lake stratification.

Sampling

Water samples were obtained with a nonmetallic Van Dorn vertical water sampler. The sampler was lowered slowly to the sampling depth. Samples were collected from June 4th to October 15th in 2010, June 30th to November 8th in 2011 and June 20th to October 3rd in 2012. All water samples were placed on ice for transport to the laboratory. Samples were analyzed within 24 h of collection or were stored in a cold room for a period not exceeding the American Public Health Association (APHA) recommended holding times (APHA, 2005).

Four sampling sites (A, B, C, D) were selected in the impoundment based on location with respect to the system and depth variability (Fig. 1). Site A was located in the deepest part of the impoundment near the air diffuser; Site B was also near the air diffuser, but further from site A; sites C and D were away from the air diffuser and located in the western and the eastern arm of the impoundment. Both sites A and B are deep while sites C and D are shallow. Site A is the deepest location while site C is the shallowest location and site D is located in the inlet.

In 2010, water quality monitoring and sampling were carried out under the conditions of aeration at sites A, B, C and D on a biweekly basis. In 2011, samples were collected on a biweekly basis during an aerated period and weekly during the period without aeration (from July 13th to September 1st) at sites A, B, C and D. In 2012, sampling was carried out under the conditions of aeration during the sampling period, except for a few days when the aeration system did not work properly.

Water samples were taken at three depths (0.5 m from the surface, Secchi depth and 1.5 m from the bottom) for chl *a* analysis at each site in 2010, in 2011 and in 2012. In addition, Secchi depth was measured using a Secchi disk at all four locations in 2010, 2011 and 2012. Total and dissolved nitrogen (N) and phosphorus (P) were also measured at same sampling depths of chl *a* at each site in 2010, 2011 and 2012.

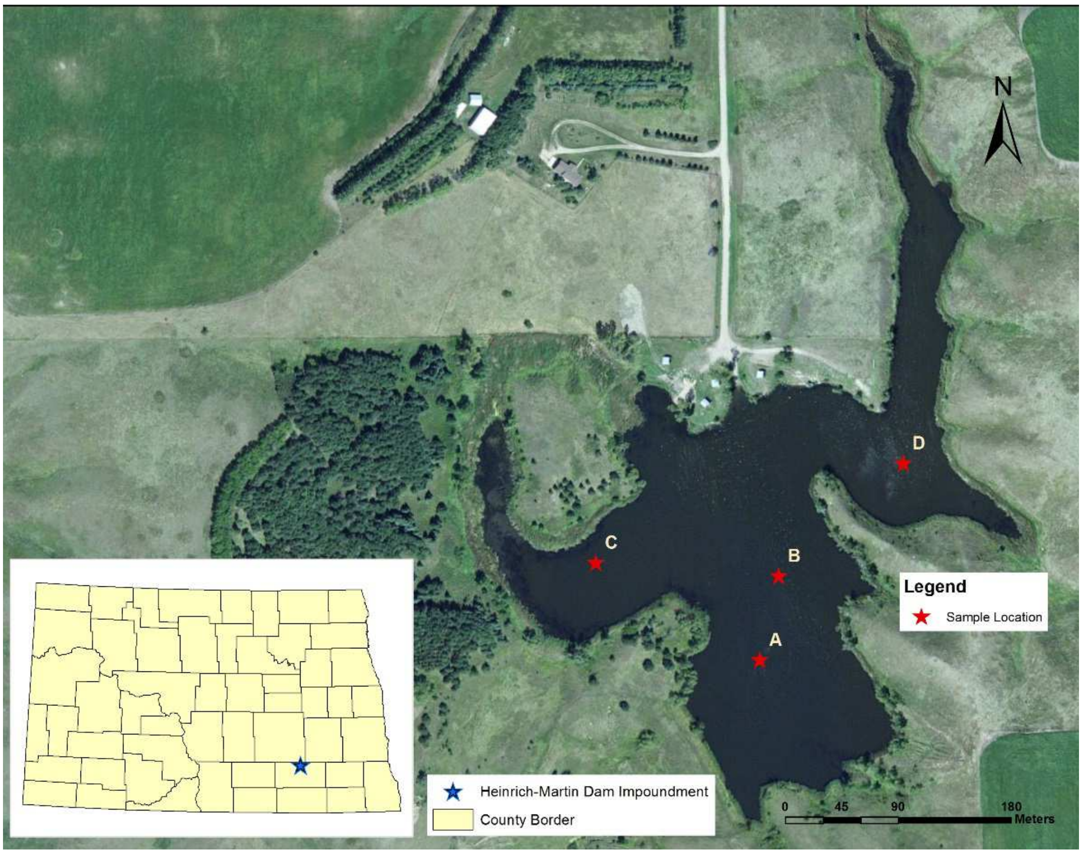


Fig. 1. Map of HMDI showing sampling locations A, B, C and D. Location of aeration system is near site A. The inlet is located in the north side, which is in the narrow gully

Chlorophyll a Analysis

For chlorophyll *a* analysis, water samples were filtered through glass-fiber filters-(pore size between 0.5 to 0.7 µm, 47 mm diameter) to separate algae from water samples. Filter papers were transferred into 20 mL bottles and 5 mL of 95% ethanol was added. Those bottles were incubated in the dark for 20 h to extract chlorophyll pigments into organic solvent (APHA, 2005). The extracted solution was filtered with glass-fiber filters to reduce the turbidity and remove any filter paper particles. Then absorbance was measured spectrophotometrically at 750 and 665 nm (Lorenzen, 1967; Sartory *et al*., 1984). All extractions were carried out in the dark to minimize degradation of chlorophyll pigments because chlorophyll could further degrade into chlorophyllides, pheophorbides and pheophytins (APHA, 2005).

Statistical Analyses

One way Analysis of Variance (ANOVA), least square mean analysis and Tukey mean comparison were used to evaluate differences in water quality parameters across the sampling depths, across the sampling dates and across the sampling locations. Furthermore, ANOVA was used to examine changes of means for the water quality parameters between non-aerated period (July and August) and aerated period. When a significant difference ($p \leq 0.05$) was found, a multiple comparison technique (least significant difference) was used to determine between which year(s) it occurred. All statistical tests were performed using Statistical Analysis System (SAS) 9.3 version and statistical significance was considered to be $p \leq 0.05$ (Cowell *et al*., 1987; Soltero *et al*., 1994; Hanson and Austin, 2012).

Results and Discussion

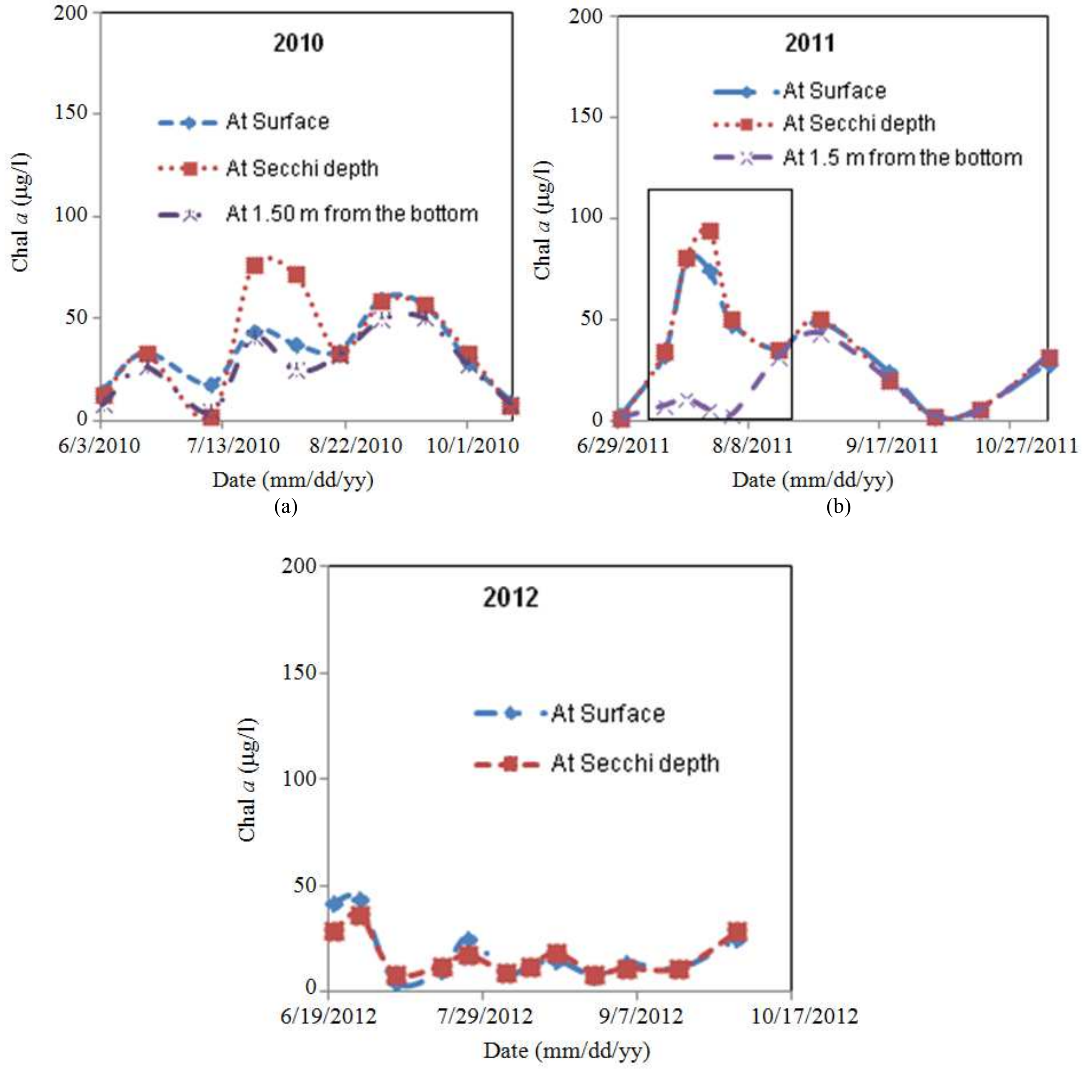
Influence of Artificial Circulation on Total Algal Biomass (as Chlorophyll a)

Chlorophyll *a* is the primary photosynthetic pigment, so a measure of its concentration in a water sample is representative of algal biomass. Figure 2 shows the variation of chlorophyll *a* concentration at site A from 2010 to 2012. In 2010, average chlorophyll *a* concentration at site A throughout the water column from July and August varied between 7.5 and 54 µg L^{-1} (Fig. 2a); however, in 2010, since the water column was vertically mixed by aeration, phytoplankton distributed throughout the water column homogeneously (Fig. 2a). No significant differences ($p > 0.05$) were observed in chlorophyll *a* concentration across depths. In 2011, however, conditions were non-aerated and chlorophyll *a* at site A showed depth variability (Fig. 2b). For instance, chlorophyll *a* concentration at both surface and Secchi depth increased gradually from 30 to 90 µg L^{-1} over time; however, concentrations rapidly dropped by 50 µg L^{-1} at both depths by the end of the non-aerated conditions (Fig. 2b). In addition, phytoplankton biomass was minimal in near bottom layers and chlorophyll *a* was between 3 and 7.5 µg L^{-1} (Fig. 2b). Furthermore, in 2011, results indicated that phytoplankton were not able to maintain their growth under non-aerated conditions because the phytoplankton population rapidly declined when nutrients were limiting. In comparison, in 2012, chlorophyll *a* concentration at site A was less than 2010 and ranged between 7 and 20 µg L^{-1} (Fig. 2c).

Dissolved inorganic nitrogen and phosphorus (ammonium-nitrogen, nitrate-nitrogen, nitrite-nitrogen and soluble reactive phosphorus) were shown to accumulate in near-bottom layers during the non-aerated period and these were not directly available for algae in the epilimnion; therefore, under non-aerated conditions in 2011, nutrients were limited for phytoplankton biomass. Further, chlorophyll *a* result likely indicated decreased total algal biomass due to lack of nutrients and competition during the stratified period of the lake (Fig. 2b).

Similar to site A, site B, in 2010, showed chlorophyll *a* ranging from 7 to 60 µg L^{-1} (Fig. 3a) and in 2011, both at the surface and Secchi depth chlorophyll *a* concentration increased from 20 to 90 µg L^{-1} during non-aeration (Fig. 3b); however, in 2012, chlorophyll *a* at site B was similar to site A (Fig. 3c). Shallow area sites C and D are shown in Fig. 4 and 5. In 2010, at site C, chlorophyll *a* was between 28 and 100 µg L^{-1} (Fig. 4a), which was higher than sites A and B. In 2012, chlorophyll *a* concentration at site C was between 10 and 34 µg L^{-1} (Fig. 4c). In contrast to sites A, B and C, at site D chlorophyll *a* was quite high (ranged between 13 and 160 µg L^{-1} at the surface in 2010 (Fig. 5a). While in 2011, chlorophyll *a* concentration was similar to sites A, B and C and algal biomass peaked at both surface and Secchi depth compared to near bottom layers (Fig. 5b). Homogenous distribution of chlorophyll *a* concentration was observed among sampling locations in 2010 and 2012 under aeration; however, in 2011, under non-aerated conditions, chlorophyll *a* showed depth variability at all four sampling locations.

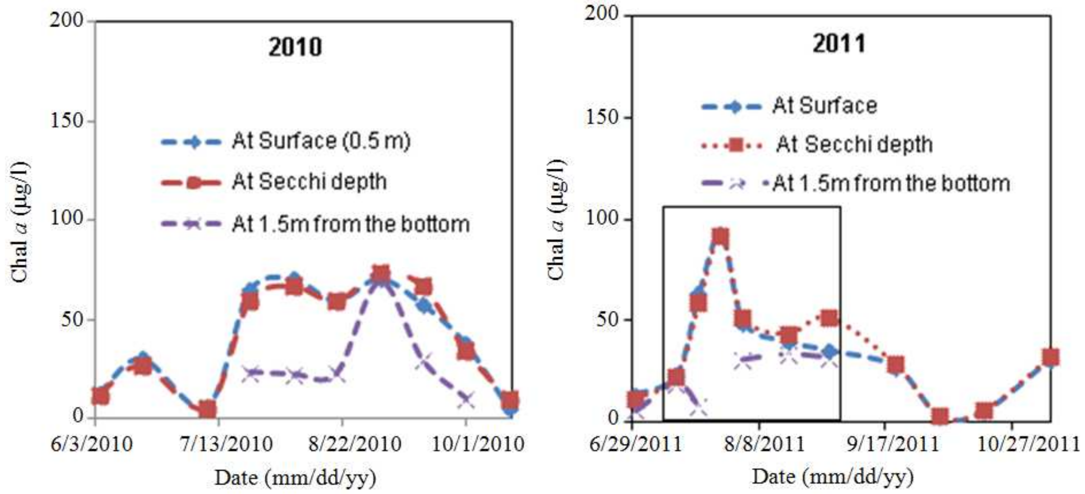
The mean chlorophyll *a* concentration at both surface and Secchi depth in 2010 was higher than chlorophyll *a* concentration in 2011, under non-aerated conditions at all locations; however, only site A showed a significant increase. In addition, the mean chlorophyll *a* concentration in 2012, was significantly lower ($p < 0.05$) at all sampling locations compared to 2011. Overall, in 2012 mean chlorophyll *a* concentration was significantly lower ($p < 0.05$) at both surface and Secchi depth at sites A, B, C and D. Therefore, artificial aeration decreased the mean chlorophyll *a* concentration significantly from 2010 to 2012 at all sampling locations.



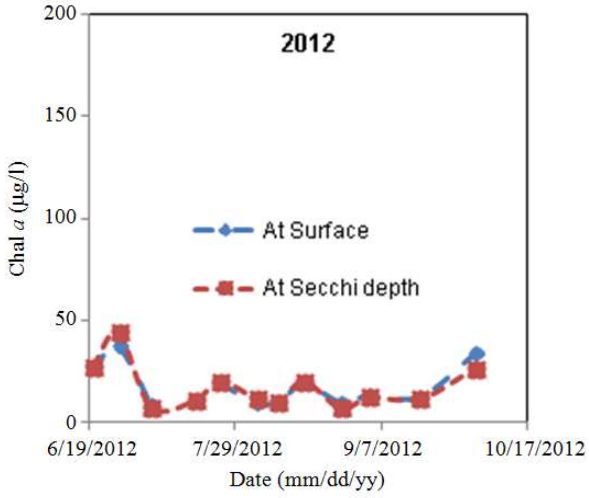
(a) (b)

(c)

Fig. 2. Chlorophyll *a* at site A: (a) in 2010, (b) in 2011 and (c) in 2012. The period without aeration is indicated by a square

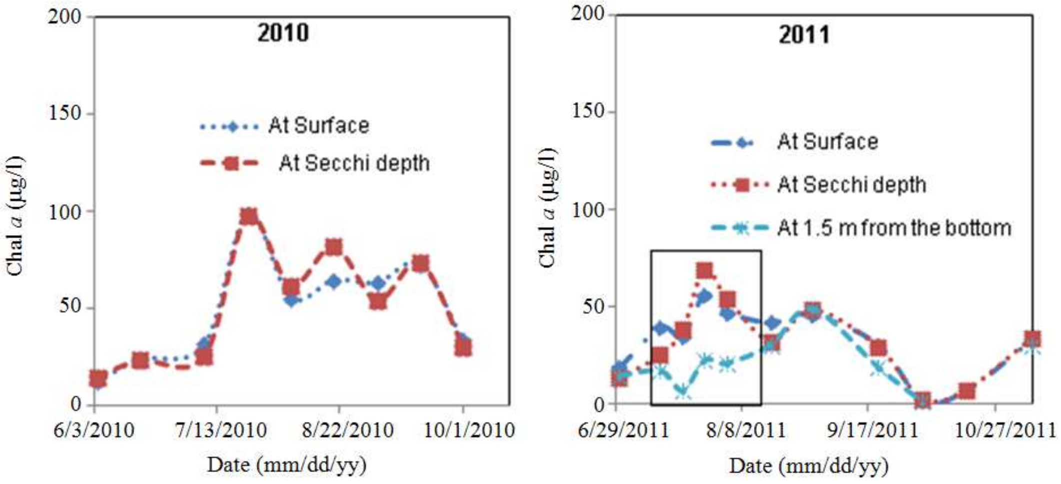


(a) (b)

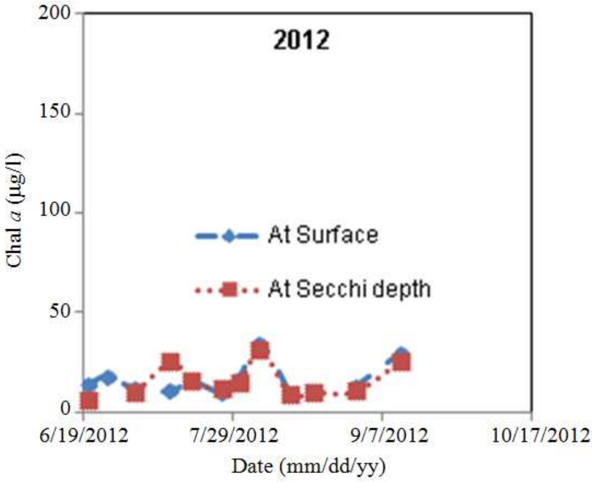


(c)

Fig. 3. Chlorophyll *a* at site B: (a) in 2010, (b) in 2011 and (c) in 2012. The period without aeration is indicated by a square

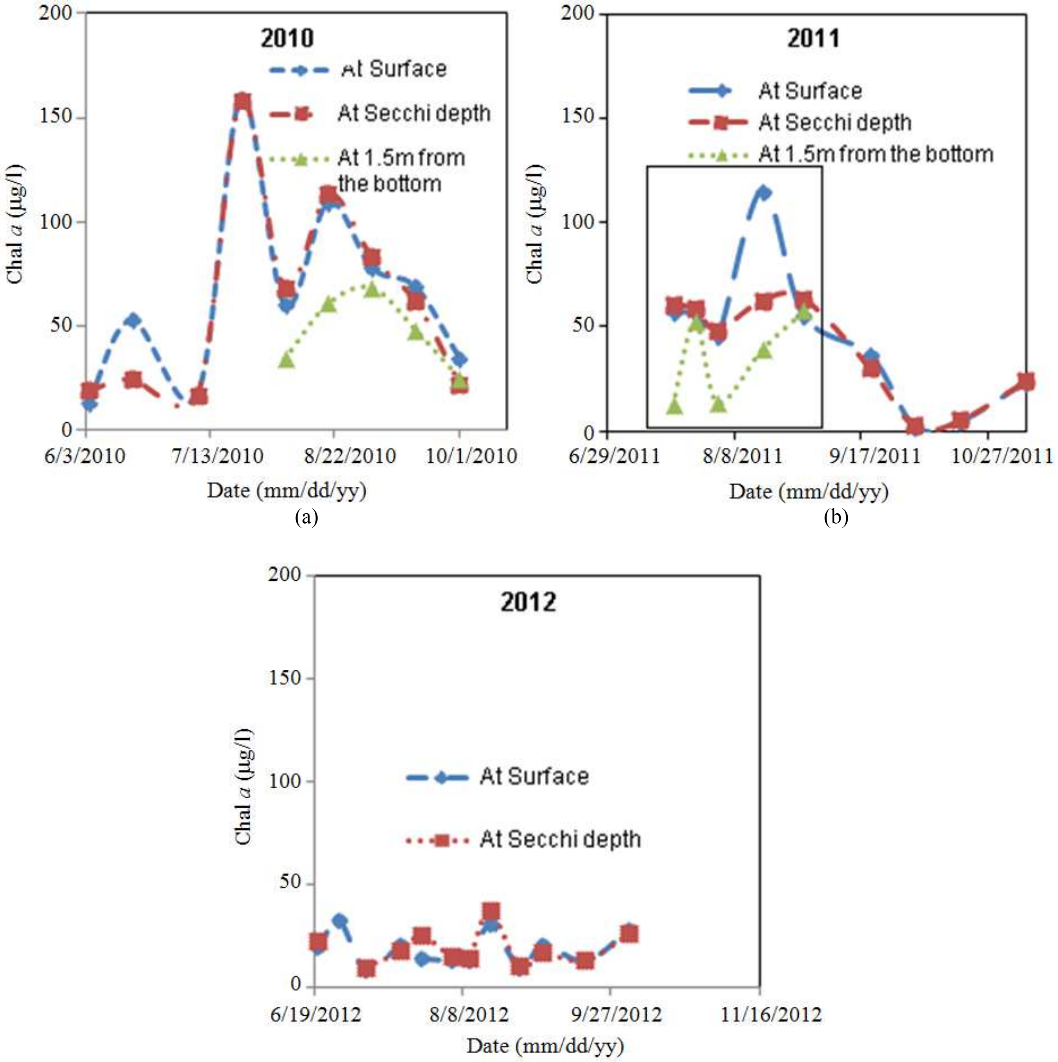


(a) (b)



(c)

Fig. 4 Chlorophyll *a* at site C: (a) in 2010, (b) in 2011 and (c) in 2012. The period without aeration is indicated by a square



(a)

(b)

(c)

Fig. 5 Chlorophyll *a* at site D: (a) in 2010, (b) in 2011 and (c) in 2012. The period without aeration is indicated by a square

Based on lake water quality data, the dominant algae group at the HMDI was cyanobacteria under non-aeration conditions; therefore, a possible explanation for suppression of algal biomass under aerated conditions could be a shift of the dominant algae group. For instance, a number of studies have documented the decline of cyanobacteria and an increase in abundance of dinoflagellates and green algae with aeration in eutrophic lakes (Cooke *et al*., 2005; Burkholder, 2001). Favorable conditions for cyanobacterial blooms in lakes include calm stable water, high solar radiation and high water temperature (Havens *et al*., 2003). Since many filamentous cyanobacteria are positively buoyant under non-aeration, the stable water column allows them to migrate to the surface to receive enough sunlight; however, with aeration, the entire water column is mixed, preventing these algae from remaining near the water surface resulting in less growth due to low irradiance (Havens *et al*., 2003). Other algal groups, such as dinoflagellates, become dominant with aeration because of their negatively buoyant capabilities, which are due to their heavy silica frustule (Reynolds, 1999).

Wax *et al*., (2011) collected water quality data from 76 reservoirs and impoundments in the rangeland plains ecological region of North Dakota between 1991 and 2011 and found that regional average chlorophyll *a* was 19.6 µg L^{-1}. The recommended Total Maximum Daily Load (TMDL) of chlorophyll *a* in North Dakota water is 20 µg L^{-1} (NDDH, 2010). In this study, mean chlorophyll *a* concentrations at both surface and Secchi depth in July and August at all four sampling locations under both aerated and non-aerated conditions in 2010 and 2011 were greater than both TMDL and the regional average. However, in 2012, mean chlorophyll *a* concentration in July and August was between 12 and 17 µg L^{-1} at all sampling locations under aerated conditions. Furthermore, in 2010, at deep locations, mean chlorophyll *a* concentrations (35 and 38 µg L^{-1}) were less than non-aerated conditions (50 and 58 µg L^{-1}) in 2011; however, in 2010, under aerated conditions, at shallow locations, mean chlorophyll *a* concentrations (65 and 80 µg L^{-1}) were higher than non-aerated conditions (44 and 65 µg L^{-1}).

Influence of Aeration on Water Clarity and its Effect on Chlorophyll a Concentration

Secchi Depth

The results of this study show the mean Secchi depth at sites A, B, C and D (1.6, 1.5, 1.2 and 1.25 m, respectively) under aerated conditions in 2010. Under non-aerated conditions, in 2011 sites A, B, C and D were 1.1, 1.2, 1.2 and 1.2 m, respectively. In contrast, in 2012, Secchi depth was greater than 2 m at all sampling locations under aerated conditions. These results indicate that under aerated conditions in 2012, lake water quality improved to the level of EPA standards for lakes and reservoirs (2 m) (EPA, 2010).

The mean Secchi depth decreased from 2010 to 2011 but not significantly at all sampling locations; however, from 2011 to 2012, the mean Secchi depth increased significantly ($p<0.05$) at all locations. Comparing the mean Secchi depth from 2010 to 2012, results indicated that the HMDI water quality improved by the artificial aeration system and the mean Secchi depth was significantly higher in 2012 than in 2010.

Secchi depth values vary as algal populations increase and decrease and Secchi depth not always followed the chlorophyll *a* concentration (Fig. 6). The Secchi depth is an indicator of water clarity; however, it is not always an indicator of trophic state. Water clarity is controlled by some other factors such as, variable amounts of chlorophyll among algal species and variable amounts of chlorophyll within an algal species as a function of cell age. In addition, nutrition level, contribution of other suspended particles, the volume versus number of phytoplankton cells and variable chlorophyll extraction efficiency as a function of cell age and solvents used also determine the water clarity (Lind, 1986; Lee *et al*., 1995; Mazumder and Havens, 1998).

Among those factors, most likely the main factor affecting the relationship between chlorophyll *a* and Secchi depth at the HMDI was volume versus number of phytoplankton cells because based on lake water quality conditions at HMDI the dominant phytoplankton genera in summer were *Aphanizomenon* sp, *Microcystis* sp and *Anabaena* sp. The number of cells per volume in each genus are different due to their different morphological features, such as *Aphanizomenon s*p. are filamentous and united to form plate-like bundles and flakes of parallel trichomes while *Microcystis* are sedentary colony of numerous spherical cells in copious mucilage forming globes and affecting for the Secchi depth turbidity (Prescott, 1982).

In addition to Secchi depth, total suspended solids also used as an indicator of water clarity (Fig. 7). Suspended particles dissipate light, which affects the depth at which algae can grow and Total Suspended Solids (TSS) include all suspended particles, such as phytoplankton, zooplankton, decaying plant and animal matter and silt in the water column. Secchi depth varied not only because of phytoplankton biomass measured by chlorophyll *a* concentration but also turbidity in the water column due to all suspended particles measured as TSS (Fig. 7).

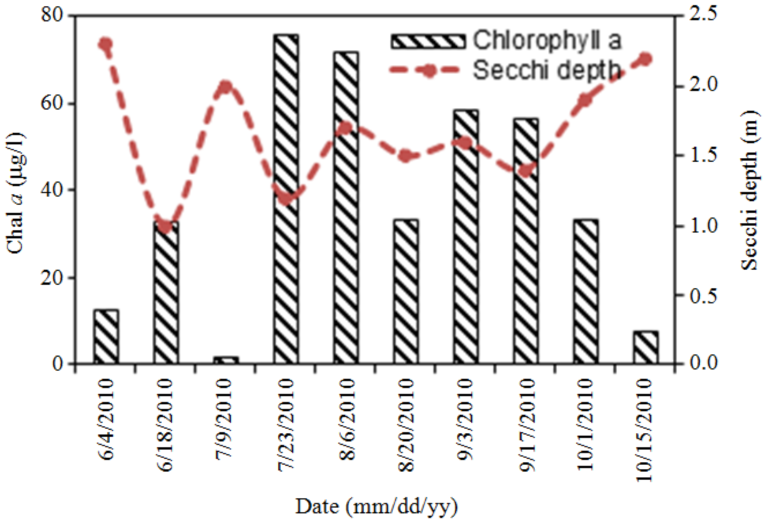


Fig. 6. Relationship between chlorophyll *a* concentration at Secchi depth level and lake's Secchi depth at site A (the deepest location) in 2010

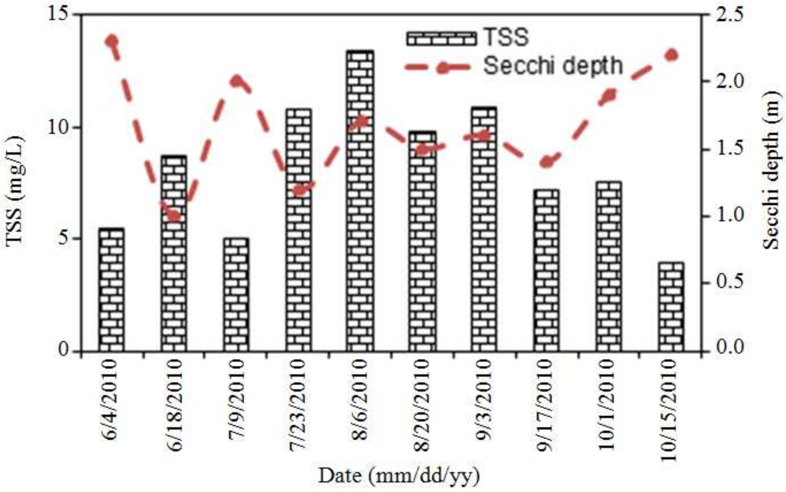


Fig. 7. Relationship between TSS at Secchi depth level and lake's Secchi depth at site A (the deepest location) in 2010

The results of this study were consistent with reports of aeration suppression of cyanoacterial blooms (Jungo *et al*., 2001; Cowell *et al*., 1987). Current results showed that when dissolved inorganic nutrient concentrations decreased at both the surface layer and Secchi depth level during the non-aerated conditions, algal biomass also dropped rapidly. During the non-aerated conditions, the majority of bioavailable N and P was near the bottom layers and growth of phytoplankton was suppressed due to lack of dissolved nutrients at the surface water column, thus changes of chlorophyll *a* depended on limiting nutrients. In addition, these result indicate that there was no external loading of dissolved inorganic nutrients into the water column under non-aerated period. If external loading of dissolved nutrients dominated during non-aerated period in 2011, algae population may not be suppressed at the HMDI.

Overall, the mean chlorophyll *a* at the HMDI decreased significantly ($p<0.05$) with the aeration in both years (2010 and 2012). Therefore, aeration suppressed total algal biomass by the end of the whole study period. Based on lake water quality, dominant algae group was cyanobacteria; therefore, a possible explanation for suppression of algal biomass under aerated conditions could be shift of dominant algae group. For instance, a number of studies have documented the decline of abundance of cyanabateria and dinoflagellates and green algae were dominant with aeration in eutrophic lakes (Cooke *et al*., 1993; Burns, 1994; Kortmann *et al*., 1994). Favorable conditions for cyanobacterial blooms in lakes include calm stable water, high solar radiation and high water temperature (Visser *et al*., 1996). Since many filamentous cyanobacteria are positively buoyant under non-aerated conditions the stable water column allows them to migrate to surface to receive enough sunlight; however, with aeration the entire water column is mixed, preventing these algae from remaining near the water surface resulting in less growth due to experience of low irradiance (Havens *et al*., 1998). Other algal groups, such as dinoflagellates, become dominant with aeration due to their negatively buoyant capabilities due to their heavy silica frustule (Reynolds, 1984). The Secchi depth, turbidity and TSS were highly varied and depended on the contribution of phytoplankton; however, conditions improved by 2012 under the conditions of aeration.

Conclusion

The study reveals that the intermittent artificial circulation in this impoundment was effective in reducing total algal biomass. This information will be useful for state departments develop numeric water quality criteria for chlorophyll *a* to protect states' waters. The circulation system has been shown to be capable of breaking down the thermal stratification that develops in the summer. Intermittent artificial circulation was successful in reducing total algal biomass (as chlorophyll *a*) throughout the water column and no spatial gradient of chlorophyll *a* was observed among sampling locations in the impoundment. Therefore, operation of the aerator in summer indicated that water quality can be greatly enhanced in terms of reduction of total algal biomass. In addition, total algal biomass was dropped to the less than recommended concentration by NDDoH, which is 20 µg L^{-1}, at all sampling locations in the HMDI impoundment. Findings of this study will help manage the HMDI impoundment more effectively under aerated and non-aerated conditions.

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Author's Contributions

Anusha Balangoda: The graduate student, who conducted experiments, collected and analyzed data and wrote the manuscript.

Shafiqur Rahman and G. Padmanabhan: Co-supervised the graduate student, reviewed the manuscript and contributed in interpreting data.

Ethics

This manuscript has neither been published, nor under consideration for publication or in press elsewhere concurrently.

References

APHA, 2005. Standard methods for the examination of water and wastewater. American Public Health Association, American Water Works Association, Water Environmental Federation.

Barbiero, R.P., B.J. Speziale and S.L. Ashby, 1996. Phytoplankton community succession in a lake subjected to artificial circulation. Hydrobiologia, 331: 109-120. DOI: 10.1007/BF00025412

Becker, A., A. Herschel and C. Wilhelm, 2006. Biological effects of incomplete destratification of hypertrophic freshwater reservoir. Hydrobiologia, 559: 85-100. DOI: 10.1007/s10750-005-4428-3

Berman, A. and B. Shteinman, 1998. Phytoplankton development and turbulent mixing in Lake Kinneret (1992-1996). J. Plankton Res., 20: 709-726. DOI: 10.1093/plankt/20.4.709

Boyce, D.G., M.L. Lewis and B. Worm, 2010. Global phytoplankton decline over the past century. Nature, 466: 591-596. DOI: 10.1038/nature09268

Boyle, K. and R. Bouchard, 2003. Water quality effects on property prices in Northern New England. LakeLine, 23: 24-27.

Brandt, P., D.J. Abson, D.A. DellaSala, R. Feller and H.V. Wehrden, 2014. Multifunctionality and biodiversity: Ecosystem services in temperate rainforests of the Pacific Northwest, USA. Biol. Conservat., 169: 362-371. DOI: 10.1016/j.biocon.2013.12.003

Burgi, H. and P. Stadelmann, 2002. Change of phytoplankton composition and biodiversity in Lake Sempach before and during restoration. Hydrobiologia, 469: 33-48. DOI: 10.1023/A:1015575527280

Burkholder, J.M., 2001. Eutrophication and Oligotrophication. In: Encyclopedia of Biodiversity, Levin, S.A. (Ed.), Academic Press, San Diego, ISBN-10: 0122268652, pp: 649-670.

Burns, F.L., 1994. Case study: Blue-green algal control in Australia by year-round automatic aeration. Lake Reservoir Manage., 10: 61-67. DOI: 10.1080/07438149409354175

Cooke, G.D., E.B. Welch, S.A. Peterson and P.R. Newroth, 1993. Restoration and Management of Lakes and Reservoirs. 2nd Edn., CRC Press, ISBN-10: 0873713974, pp: 560.

Cooke, G.D., E.B. Welch, S.A. Peterson and S.A. Nichols, 2005. Restoration and Management of Lakes and Reservoirs. 3rd Edn., CRC Press, ISBN-10: 1420032100, pp: 616.

Cowell, B.C., C.J. Dawes, W.E. Gardiner and S.M. Scheda, 1987. The influence of whole lake aeration on the limnology of a hypereutrophic lake in central Florida. Hydrobiologia, 148: 3-24. DOI: 10.1007/BF00018162

Dziuk, H. and S. Heiskary, 2003. Local economic effects of healthy lakes. LakeLine, 23: 21-24.

EPA, 2010. Ambient water quality criteria recommendations: Information supporting the development of state and tribal nutrient criteria (Lakes and Reservoirs in Nutrient Ecoregion IV). United States Environmental Protection Agency.

Everard, M. and R. McInnes, 2013. Systemic solutions for multi-benefit water and environmental management. Sci. Total Environ., 461-462: 170-179. DOI: 10.1016/j.scitotenv.2013.05.010

Fedrowitz, K., J. Koricheva, S.C. Baker, D.B. Lindenmayer and B. Palik *et al.*, 2014. Can retention forestry help conserve biodiversity? A meta-analysis. J. Applied Ecol., 51: 1669-1679. DOI: 10.1111/1365-2664.12289

Hannon, M., J. Gimpel, M. Tran, B. Rasala and S. Mayfield, 2010. Biofuels from algae: Challenges and potential. Biofuels, 1: 763-784. DOI: 10.4155/bfs.10.44

Hanson, D. and D. Austin, 2012. Multiyear destratification study of an urban, temperate climate, eutrophic lake. Lake Reservoir Manage., 28: 107-119. DOI: 10.1080/07438141.2012.671229

Havens, K.E., R.T. James, T.L. East and V.H. Smith, 2003. N:P ratios, light limitation and cyanobacterial dominance in a subtropical lake impacted by non-point source nutrient pollution. Environ. Pollut., 122: 379-390. PMID: 12547527

Havens, K.E., E.J. Phlips, M.F. Cichra and B.L. Li, 1998. Light availability as a possible regulator of cyanobacteria species composition in a shallow subtropical lake. Fresh Water Biol., 39: 547-556. DOI: 10.1046/j.1365-2427.1998.00308.x

Hyenstrand, P., P. Blomqvist and A. Pettersson, 1998. Factors determining cyanobacterial success in aquatic systems: A literature review. Arch. Hydrobiol. Spec. Issue Adv. Limnol., 51: 41-62.

Jungo, E., P.M. Visser, J. Stroom and L.R. Mur, 2001. Artificial mixing to reduce growth of the blue-green alga Microcystis in Lake Nieuwe Meer, Amsterdam: An evaluation of 7 years of experience. Water Sci. Technol. Water Supply, 1: 17-23.

Kasprzak, P., J. Padisak, R. Koschel, L. Krienitz and F. Gervais, 2008. Chlorophyll a concentration across a trophic gradient of lakes: An estimator of phytoplankton biomass? Limnologica Ecol. Manage. Inland Waters, 38: 327-338. DOI: 10.1016/j.limno.2008.07.002

Kortmann, R.W., G.W. Knoecklein and C.H. Bonnell, 1994. Aeration of stratified lakes: Theory and practice. Lake Reservoir Manage., 8: 99-120. DOI: 10.1080/07438149409354463

Lee, G.F., A. Jones-Lee and W. Rast, 1995. Secchi depth as a water quality parameter. Report of G. Fred Lee and Associates, EI Macero, CA.

Lind, O.T., 1986. The effect of non-algal turbidity on the relationship of Secchi depth to chlorophyll *a*. Hydrobiologia, 140: 27-35. DOI: 10.1007/BF00006726

Lilndenschmidt, K.E., 1999. Controlling the growth of *Microcystis* using surged artificial aeration. Int. Rev. Hydrobiol., 84: 243-254. DOI: 10.1002/iroh.199900026

Lorenzen, C.J., 1967. Determination of chlorophyll and pheo-pigments: Spectrophotometric equations. Limnol. Oceanography, 12: 343-346. DOI: 10.4319/lo.1967.12.2.0343

Mazumder, A. and K.E. Havens, 1998. Nutrient-chlorophyll-Secchi relationships under contrasting grazer communities of temperate versus subtropical lakes. Canadian J. Fisheries Aquat. Sci., 55: 1652-1662. DOI: 10.1139/f98-050

NDDH, 2010. North Dakota 2010 integrated section 305(b) water quality assessment report and section 303(d) list of waters needing total maximum daily loads. North Dakota Department of Health, Division of Water Quality.

Overmoe, K.A., 2008. Influence of artificial aeration on water quality at Heinrich-martin impoundment, North Dakota. Thesis, North Dakota State University.

Pastrok, R.A., T.C. Ginn and M.W. Lorenzen, 1982. Review of Aeration/Circulation for Lake Management. In: Restoration of Lakes and Inland Waters, EPA 440/5-81-010. U.S. Environmental Protection Agency, Washington, D.C., pp: 124-133.

Prescott, G.W., 1982. Algae of the Western Great Lakes Area: With an Illustrated Key to the Genera of Desmids and Freshwater Diatoms. 1st Edn., Otto Koeltz Science Publishers, Koenigstein, ISBN-10: 3874292053, pp: 977.

Ramaraj, R., D.W. David and P.G. Chen, 2013. Chlorophyll is not accurate measurement for algal biomass. Cha. Mai J. Sci., 40: 547-555.

Reynolds, C.S., 1984. The Ecology of Freshwater Phytoplankton. Cambridge University Press, Cambridge, UK, ISBN-10: 0521282225, pp: 384.

Reynolds, C.S., 1999. Non-determinism to probability, or N: P in the community ecology of phytoplankton: Nutrient ratios. Arch. Hydrobiol., 146: 23-35.

Sartory, D.P. and J.U. Grobbelaar, 1984. Extraction of chlorophyll a from freshwater phytoplankton for spectrophotometric analysis. Hydrobiologia, 114: 177-187. DOI: 10.1007/BF00031869

Schladow, S.G. and I.H. Fisher, 1995. The physical response of temperate lakes to artificial destratification. Limnol. Oceaonography, 40: 359-373. DOI: 10.4319/lo.1995.40.2.0359

Sgroi, F., A.M.D. Trapani, M. Fodera, R. Testa and S. Tudisca, 2015. Economic assessment of *Eucalyptus* (spp.) for biomass production as alternative crop in Southern Italy. Renewable Sustainable Energy Rev., 44: 614-619. DOI: 10.1016/j.rser.2015.01.032

Sherman, B.S., J. Whittington and R.L. Oliver, 2000. The impact of artificial destratification on water quality in Chaffey Reservoir. Arch. Hydrogiol. Spec. Issues Adv. Limnol., Limnol. Lake Manage., 55: 15-29.

Simmons, J., 1998. Algal control and destratification at Hanningfield Reservoir. Water Sci. Technol., 37: 309-316. DOI: 10.1016/S0273-1223(98)00038-9

Soltero, R.A., L.M. Sexton, K.I. Ashley and K.O. Mckee, 1994. Partial and full lift hypolimnetic aeration of Medical Lake, WA to improve water quality. Water Res., 28: 2297-2308. DOI: 10.1016/0043-1354(94)90045-0

Steinberg, C., 1983. Effects of artificial destratification on the phytoplankton populations in a small lake. J. Plankton Res., 5: 855-864. DOI: 10.1093/plankt/5.6.855

Thiamdao, S., M. Motham, J. Pekkoh, L. Mungmai and Y. Peerapornpisal, 2012. Nostochopsis lobatus Wood em. Geitler (Nostocales), edible algae in Northern Thailand. Chiang Mai J. Sci., 39: 119-127.

Trapani, A.M.D., R. Squatrito, M. Fodera, R. Testa and S. Tudisca *et al*., 2014. Payment for environmental services for the sustainable development of the territory. Am. J. Environ. Sci., 10: 480-488. DOI: 10.3844/ajessp.2014.480.488

Visser, P., B. Ibelings, B.V.D. Veer, J. Koedood and R. Mur, 1996. Artificial mixing prevents nuisance blooms of the cyanobacterium *Microcysti*s in Lake Nieuwe Meer, the Netherlands. Freshwater Biol., 36: 435-450. DOI: 10.1046/j.1365-2427.1996.00093.x

Wax, P., A. McDonald and M. Ell, 2008. North Dakota LAKE water quality assessment reports. North Dakota Department of Health, Division of water quality.

Wax, P., A. McDonald and M. Ell, 2011. North Dakota lake water quality assessment reports. North Dakota Department of Health.

Wetzel, R.G., 2001. Limnology: Lake and River Ecosystems. 1st Edn., Gulf Professional Publishing, San Diego, ISBN-10: 0127447601, pp: 1006.