Hydrobiologia
DOI 10.1007/s10750-009-9938-y

LAKE RESTORATION

Does sediment capping have post-application effects on zooplankton and phytoplankton?

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Received: 18 May 2009 / Accepted: 8 September 2009

Abstract Although in situ sediment capping is frequently used to reduce internal loading of contaminants and nutrients, post-application assessment rarely includes the potential undesirable short-term effects on plankton species composition. We hypothesised that a modified zeolite (Z2G1) application as a sediment capping agent in Lake Okaro, New Zealand, could cause significant undesirable shifts in species composition of both zooplankton and phytoplankton due to burial of resting stages or interference with feeding for the zooplankton. Alternatively, we predicted that the capping agent might have no effect due to, for example, the coarse grain size of the material (1–3 mm). We used multidimensional scaling (MDS) and analysis of similarity (ANOSIM) to identify any adverse effects of Z2G1 on zooplankton and phytoplankton species composition (i.e. shifts in community structure, including species loss) by comparing the community structure before and after the Z2G1 application. We found no significant differences in species composition before and after the Z2G1 application at the depths investigated (surface and 9 m). However, all of the analyses showed statistically significant differences among seasons, indicating seasonal variations in plankton composition far outweigh those that may have resulted from the Z2G1 application. Coarse particle size, low dose rate and a restricted area where the sediment capping agent was applied were considered to be the factors limiting potential adverse effects on plankton species. Considerations of finer-grained material to increase coverage and efficacy of phosphorus adsorption require assessment for their effects on zooplankton, however, and a direct mode of application into the hypolimnion is recommended to minimise effects on zooplankton and phytoplankton communities.

Keywords Eutrophication · Lake Okaro · Modified zeolite · Turbidity

Introduction

Internal loading of phosphorus (P) and nitrogen (N) can contribute significantly to nutrient loads entering the water column of lakes (Søndergaard et al., 2003). Such inputs can delay the improvement of water quality following restoration measures (Jeppesen et al., 2005). Several methods have therefore been developed to reduce P in both the water column and bottom sediments of lakes (Klapper, 2003). Flocculation or P precipitation with iron or aluminium salts

Guest editors: D. P. Hamilton, M. J. Landman / Lake Restoration: An Experimental Ecosystem Approach for Eutrophication Control

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Published online: 23 October 2009

(Alum) is the most common method to decrease P content in the water column of lakes with long retention times (Cooke et al., 1993). Alum can form a P-adsorbing floc cap on the surface layer of sediments, or alternatively mixes within the sediments, where it continues to reduce P release to the overlying water (Cooke et al., 1993; Rydin et al., 2000; Pilgrim et al., 2007). Another technique to address internal loading is sediment capping, for which the aim is to prevent nutrients (or other contaminants) being released from the sediments (Förstner & Apitz, 2007). In this respect, sand, mineral soils and clay minerals are applied to the lake as passive capping agents to physically seal off the affected areas of the lake bottom sediment (Klapper, 2003). Sediment capping agents can also be chemically active P-inactivation agents designed to permanently bind P. These may be naturally occurring clay minerals modified to enhance P uptake capacity (Hickey & Gibbs, 2009).

Despite the frequent use of in situ capping (e.g. Yamada et al., 1987; Robb et al., 2003; Berg et al., 2004), post-application assessment has typically focused on reductions in the concentration of the targeted substance (e.g. P). Rarely have there been investigations of the potential undesirable short-term effects on, for example, plankton species composition. Naturally, species composition of phytoplankton can be influenced by a number of factors such as mixing regime, light climate and nutrient status (Ryan et al., 2006). As nutrient concentrations are reduced after in-lake restoration, potentially resulting in reduced dominance of cyanobacteria (Reynolds, 1998), this could lead to improved food quality for zooplankton (Gulati & Demott, 1997; Duggan et al., 2002a). The lack of recognition of the potential direct effects of sediment capping materials on plankton community structure is surprising, however, given some of the known acute and chronic toxic effects of flocculants such as Alum, which may be exacerbated by excessive dose rates, poor application timing or inappropriate environmental conditions (e.g. low pH; Lamb & Bailey, 1981; Gensemer & Playle, 1999). For example, applications of the flocculent Alum have been shown to lead to short term reductions in zooplankton abundance, biomass and species richness in Newman Lake, Washington, USA (Schumaker et al., 1993). An Alum application in Lake Okaro, New Zealand, led to increased ammonium concentrations in the water column, possibly as a result of zooplankton mortality (Paul et al., 2008). Randall et al. (1999) tested the acute toxicity of iron salts on daphnid zooplankton in a laboratory scale study, and found that the presence of particulate iron caused increased mortalities and reduced the number of broods per female.

The deposition of large volumes of sediment into a lake might be expected to affect zooplankton and phytoplankton species composition immediately following application in several ways. For example, experimental studies on the impacts of inorganic sediments have shown that with increasing turbidity there are reductions in the standing stocks of daphnid zooplankton (Hart, 1986) and reduced zooplankton feeding rates (Kirk, 1991). Consequently, lakes with high suspended sediment loads commonly have a limited richness of zooplankton species (e.g. Duggan et al., 2002a). Robb et al. (2003) suggested that an increase in phytoplankton growth rates following treatment with the capping agent Phoslock™ at sites in the Swan-Canning Estuary, Australia, may have been caused by a loss of grazers (i.e. zooplankton). Finally, burial of phytoplankton cysts and zooplankton diapausing eggs by the capping agent also have the potential to reduce plankton recruitment following application of these agents. With the basal positions of phytoplankton and zooplankton in aquatic food webs, alterations in these components may alter the energy flow to higher trophic levels and have ecosystem-wide effects.

In New Zealand, the water quality of many of the Te Arawa Lakes of the Rotorua region has declined significantly, coinciding with the development of surrounding catchments for agriculture (McColl & Hughes, 1981; Hamilton, 2005). Due to high external nutrient loading over many decades, the bottom sediments have become enriched with organic matter, and consequently several of the lakes have high internal nutrient loadings (White et al., 1978; Burger et al., 2007). An aluminium-modified zeolite (Z2G1; Blue Pacific Minerals, Matamata, New Zealand) application in Lake Okaro, Rotorua, was carried out in September 2007. Zeolite is a porous alumino-silicate material that has a large specific adsorptive surface area due to a fine porous structure. A naturally occurring zeolite in the Rotorua region has been modified by Scion to significantly improve the nutrient uptake capacity of the natural material. The modified zeolite was applied as a sediment capping agent with the primary purpose of not only reducing internal loading of phosphate, but also targeting ammonium. We hypothesised that the Z2G1 application may cause significant shifts in species composition of zooplankton and phytoplankton due to burial of resting stages or through interference of zooplankton grazing. Alternatively, Z2G1 might have no effect on plankton species composition due to, for example, a restricted sediment capping target area and the relatively coarse grain size of the application. The objective of this study was to identify any adverse effects of Z2G1 on zooplankton and phytoplankton species composition (e.g. shifts in community structure, including species loss) by comparing the community structure before and after the Z2G1 application in Lake Okaro.

Materials and methods

Study site

Lake Okaro (Fig. 1) is a small (0.32 km^2), shallow lake (18 m max. depth) in the Rotorua region, New Zealand. It is located 27 km south of Rotorua township and was formed c. 800 years ago from a geothermal explosion crater (Lloyd, 1959). The inflows to the lake are two unnamed streams that enter from the north-west. Both have been diverted through a 2.3 ha constructed wetland since 2006 to reduce external nutrient loads (Tanner et al., 2007). The Haumi Stream in the south-east of the lake is the only outflow (Forsyth et al., 1988). The catchment area (3.89 km^2) is now mostly used for dairy production. Lake Okaro has been eutrophic since the early 1960s (Jolly, 1977) and regular cyanobacterial blooms have persisted to this point in time (Forsyth et al., 1988; Paul et al., 2008). Z2G1 was applied between 25 and 28 September 2007. A total of 110 tonnes of the material (grain size 1–3 mm) was applied over the water surface, corresponding to the area where lake depths are greater than 5 m. This was equivalent to the 0.2 km^2 area of lakebed above which the hypolimnetic waters become anoxic during the seasonal stratification period of c. 8 months in this warm monomictic lake.

Sampling

Zooplankton samples were collected in duplicate using a 10 l Schindler–Patalas trap at site A (Fig. 1). Samples were taken monthly (26 January 2007–19 June 2008) and weekly 2 weeks before and after the Z2G1 application. Water from the trap was passed through a 40 μm mesh filter, and the retained zooplankton were preserved in 50% ethanol (final concentration). For enumeration, samples were passed through a 40 μm mesh to remove ethanol and to attain

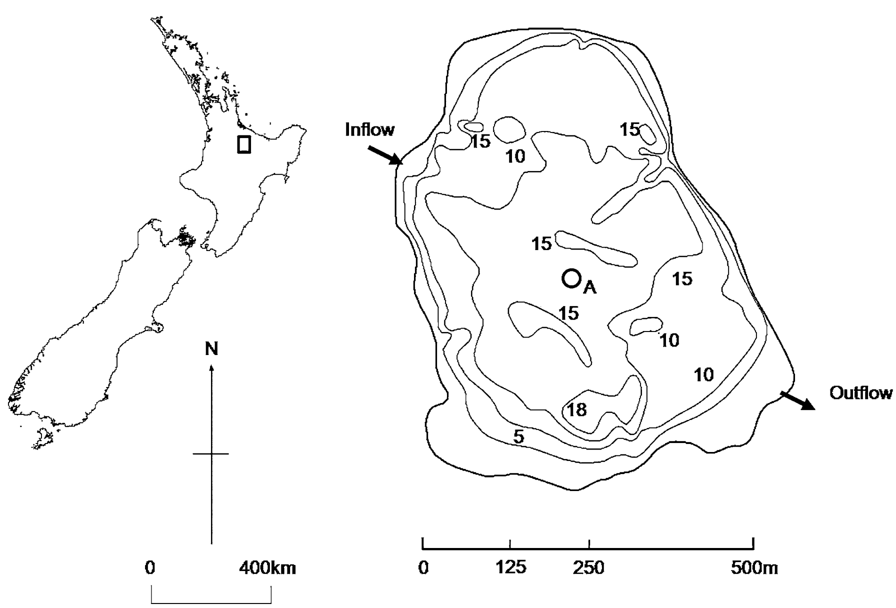

Fig. 1 Map of New Zealand and of Lake Okaro showing depth contours 5, 10, 15, and 18 m and location of lake sampling station

a final known volume of 30 and 40 ml depending on the density of algae. Samples were enumerated in 5 ml aliquots in a gridded Perspex tray until at least 300 counts were obtained, or the entire sample was enumerated. Species were identified using standard guides (e.g. Chapman & Lewis, 1976; Shiel, 1995).

Phytoplankton samples were collected using a 10 l Schindler–Patalas trap at site A (Fig. 1) from 0 to 9 m depths, and transferred to 150 ml polycarbonate jars. Samples were collected from the same days and depths as the zooplankton samples. Samples were preserved using Lugol's iodine solution and stored in the dark until analysis. Phytoplankton cell counts were carried out on settled samples in Utermöhl chambers (Utermöhl, 1958) using keys and descriptions from Prescott (1978), Cox (1996), John et al. (2003), and Baker & Fabbro (1999). Depending on cell densities, 1–3 ml aliquots were settled in the chambers and enumerated at 200–400× magnification.

Analytical methods

Non-metric multidimensional scaling (MDS) and analysis of similarities (ANOSIM) were used to detect patterns in species composition of both zooplankton and phytoplankton at each depth, and to test for differences before and after the Z2G1 application. MDS was performed on a similarity matrix based on the Bray–Curtis similarity coefficient calculated on fourth-root transformed abundance data of dominant taxa. MDS builds a 2-D "map" based on the similarities among samples as defined by the similarity matrix. A stress value is provided as a measure of the goodness of the map's fit relative to the similarity matrix. Phytoplankton and zooplankton were considered dominant if they comprised >5% of the total abundance on any date, for either of the depth combinations, over the study period. The fourth-root transformation was chosen to reduce any undue influence of highly abundant community members. To determine whether changes in taxonomic composition before and after the Z2G1 application were significant, we applied ANOSIM to the similarity matrix underlying the MDS ordination. ANOSIM is a non-parametric permutation test to examine a priori hypotheses. Sample periods were split into two groups for testing; before and after application (January 2007–September 2007 and October 2007–June 2008). ANOSIM provides a measure of dissimilarity of groups of samples in the form of an *R*-statistic that lies between 0 and 1; values close to 1 imply that the groups are dissimilar while those approaching zero are very similar. In addition, we tested for the influence of season on community composition relative to the influence of Z2G1, with sample groups divided seasonally into summer (December-February), autumn (March–May), winter (June–August) and spring (September–November). We adjusted the acceptable *P* value ($\alpha = 0.05$) using a Dunn–Šidák correction to allow for the non-independence of these tests. MDS and ANOSIM were performed using the PRIMER 6 statistical software package (PRIMER 6.1.6, Plymouth Marine Laboratory).

Results

Zooplankton dynamics

Amongst the zooplankton rotifers were numerically dominant over crustaceans during the study (Fig. 2). Nine species comprised greater than 5% of the total abundance on any date (Fig. 2). The dominant crustacean *Bosmina meridionalis* and rotifers *Pompholyx sulcata*, *Keratella cochlearis* and *Polyarthra dolichoptera* were generally found to have highest abundances in the surface waters when the lake was mixed (peaking in July), although they also showed peaks in abundance in the 9 m samples at other times (typically in summer). The calanoid copepod *Calamoecia lucasi* and rotifer *Trichocerca similis* were most abundant in winter and spring (June to November). *Hexarthra intermedia* and *Filinia longiseta* were most abundant during summer and autumn (December to May), while *Filinia novaezealandiae* had short-lived peaks of abundance in February 2007 and December 2007 at 9 m only. Overall, abundance peaks were more predictable in the surface waters than at 9 m. The application of Z2G1 did not appear to alter the dynamics of any of the dominant species.

Phytoplankton dynamics

Twenty-one phytoplankton species were found to comprise greater than 5% of any sample over the entire study period. The dominant species that comprised greater than 10% of any samples are presented in Fig. 3. *Fragilaria crotonensis* was dominant during

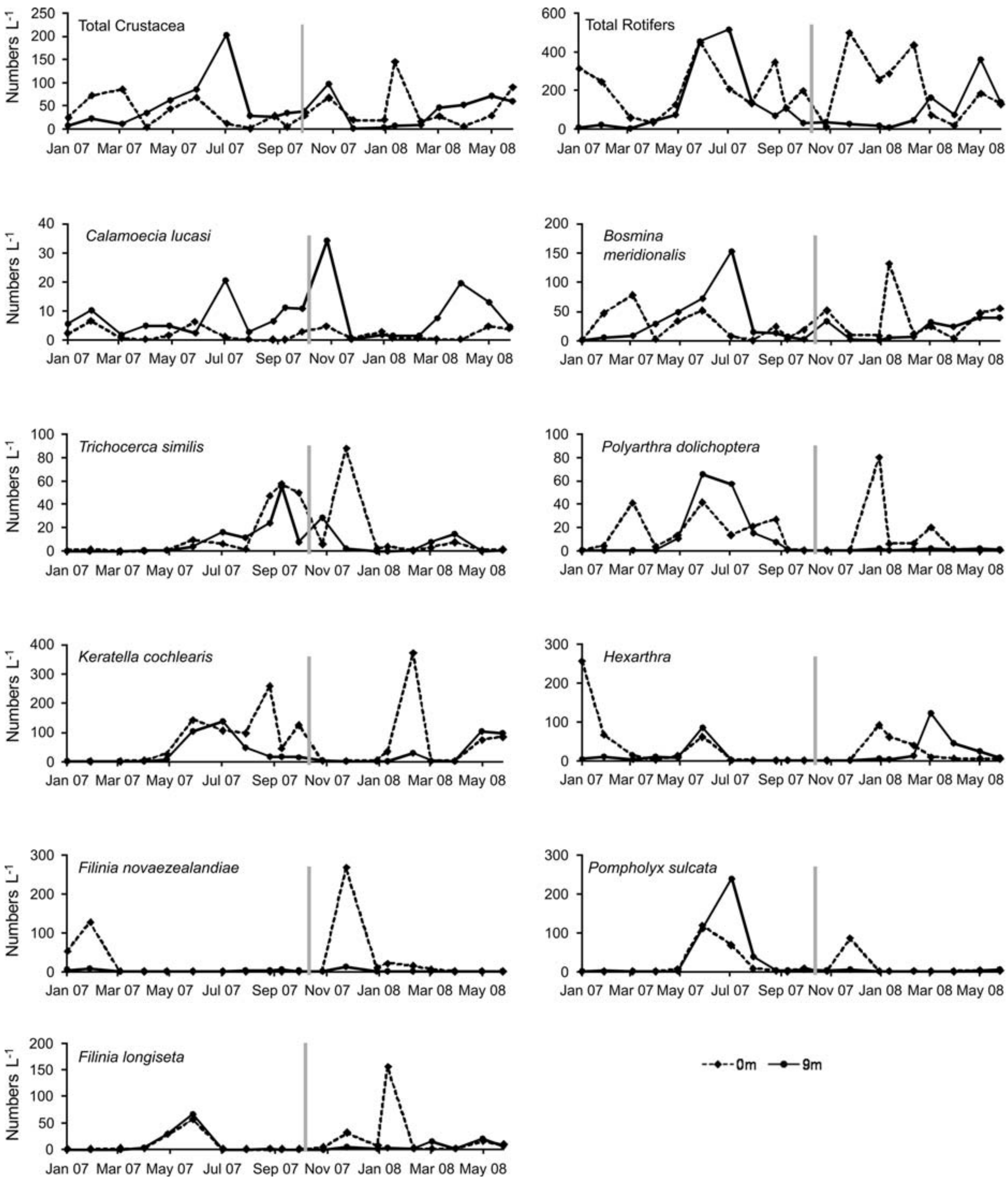

Fig. 2 Temporal and spatial distribution of total crustaceans and total rotifers and abundant zooplankton species comprising greater than 5% of any sample (numbers l^{-1}). *Shaded area* represents days of modified zeolite application

summer months (February–March) at depths 0 and 9 m. *Asterionella formosa* and *Microcystis aeruginosa* were most abundant at the end of the stratification period (May–June). *Anabaena spiroides* and *Dictyosphaerium* sp. had peaks in abundance during spring (September–October) but were almost absent for the

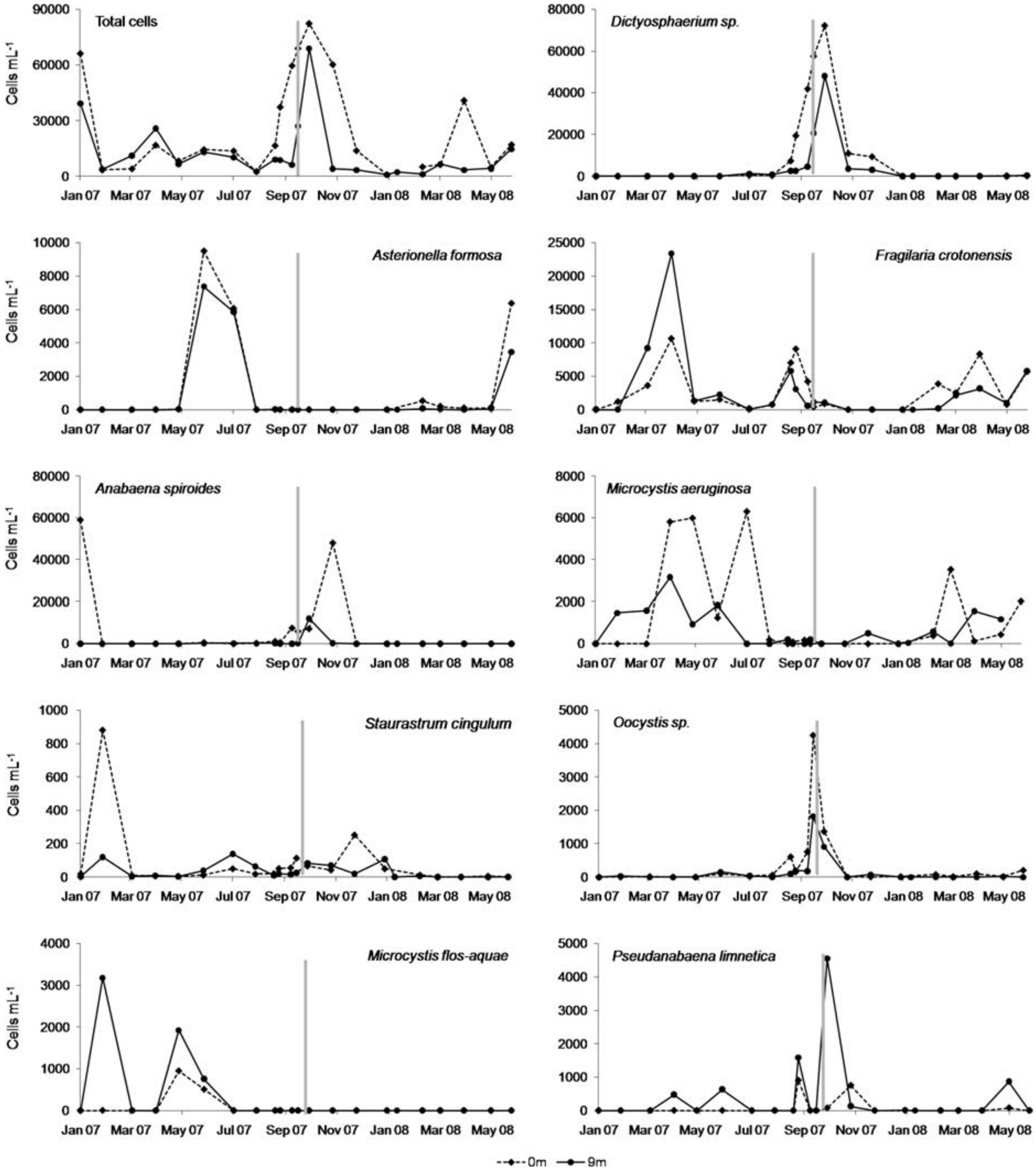

Fig. 3 Temporal and spatial distributions of total phytoplankton concentrations (cells ml^{-1}) and abundant phytoplankton species comprising greater than 10% on most sampling days. *Shaded area* represents days of modified zeolite application

remainder of the study period. *Staurastrum cingulum* varied little throughout the study period but with a peak in the 9 m sample in February 2007. *Oocystis* sp. and *Pseudanabaena limnetica* were most abundant during October 2007. *Microcystis flos-aquae* showed highest abundance during summer 2007 (January–May) but was often close to detection limits (<10 planktonic units ml^{-1}) during 2008. Green algae were the dominant taxonomic group during summer (July–January), representing up to 90% of total cell counts at 0 and 9 m depth, followed by Cyanobacteria, Bacillariophyta, Eulenophyta and Cryptophyta. The application of Z2G1 did not appear to alter the abundance of any of the dominant species or taxonomic groups.

Community analyses

The low stress values of the MDS plots (Fig. 4) indicate that the ordinations provide good representations of the relationships among samples (Clarke & Warwick, 2001). In all plots, a strong seasonal influence is apparent. Samples for each season were typically closely associated with one another, regardless of year, while there was a reasonable separation of samples from different seasons. For example, in the ordination of zooplankton at 0 m, summer (December–February) samples are located in the top left region of the ordination, autumn samples in the bottom left, and winter and spring samples in the bottom right. Samples from equivalent times in 2007 and 2008 were typically found closely associated in the MDS plot (e.g. summer and autumn samples), despite the application of Z2G1 between these dates.

Results from ANOSIM supported these findings (Table 1). All of the analyses of zooplankton and phytoplankton communities showed composition to be statistically indistinguishable before and after the Z2G1 application ($P > 0.05$). However, all of the analyses showed statistically significant differences among seasons (all $P < 0.01$), indicating seasonal variation in plankton composition far outweighed any variation caused by Z2G1 application.

Discussion

There appeared to be no discernible effect of the Z2G1 application in Lake Okaro on zooplankton or phytoplankton composition in the days or months immediately following its application. Seasonally induced

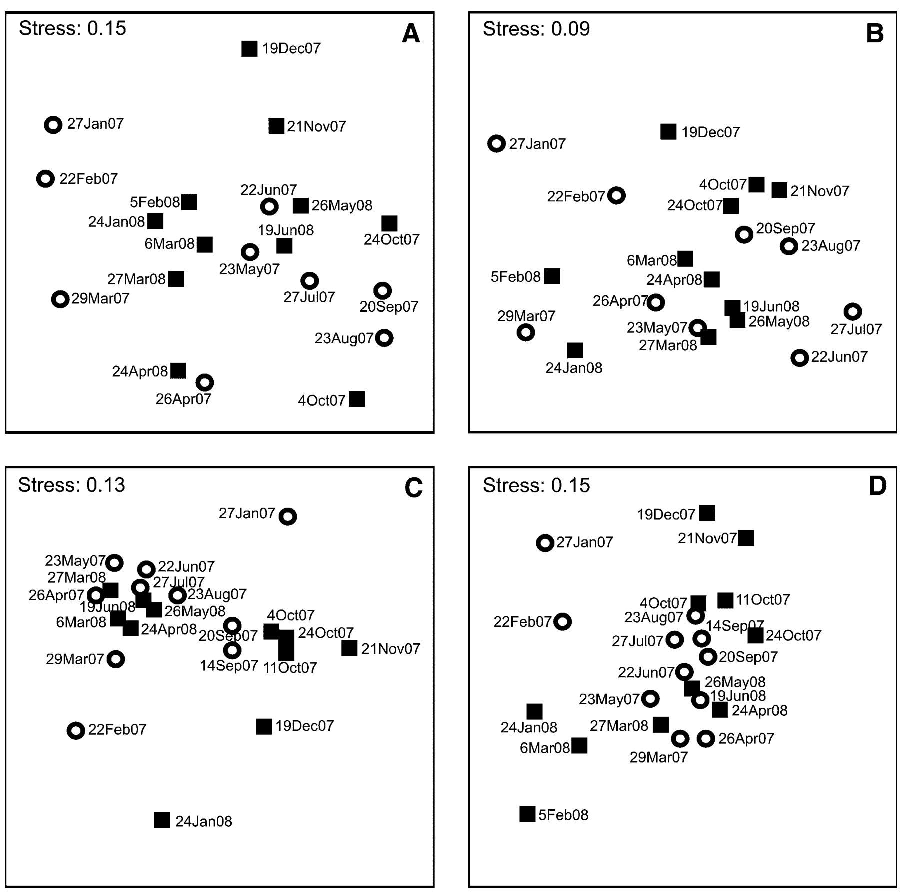

Fig. 4 Multidimensional scaling (MDS) ordinations of sampling days based on abundant taxa with **A** zooplankton at 0 m, **B** zooplankton at 9 m, **C** phytoplankton at 0 m, and **D** phytoplankton at 9 m. *Circles* signify dates before application of Z2G1, while *squares* signify dates post-application

Table 1 Results (R and P values) from ANOSIM tests for differences in zooplankton and phytoplankton community composition before and after application of Z2G1 and for seasonal change

Taxon	Treatment	0 m	9 m
Zooplankton	Application before/after	0.003 (0.67)	0.037 (0.41)
	Season	0.406 (<0.01)	0.518 (<0.01)
Phytoplankton	Application before/after	0.090 (0.16)	0.043 (0.36)
	Season	0.664 (<0.01)	0.447 (<0.01)

changes were the over-riding factor determining plankton composition during our study. The lack of effects on plankton species composition observed in the Z2G1 application to Lake Okaro indicates that it may have fewer undesirable direct (e.g. from free Al^{3+}) and indirect (e.g. from low pH) effects on biota over other methods for reducing internal nutrient loadings (e.g. where Alum application is inappropriate, because of low pH), providing it is sufficiently efficacious in removing P. Major reasons for a lack of undesirable impacts may include the coarse particle sizes (1–3 mm) of the Z2G1 material and the limited area of the lake where it was applied.

Sediment particle size is a significant factor influencing algal ingestion rates of zooplankton, and the large size of particles used in the current Z2G1 application may have reduced any negative effects on zooplankton grazing. Kirk (1991) demonstrated that coarse clay (mean particle size of 1 μm) can significantly reduce the feeding rate of *Daphnia*. In Lake Okaro, however, the sediment capping agent was applied as a coarse material (grain size 1–3 mm), larger than the size of particles ingested by zooplankton (or indeed of most zooplankton themselves). The increased settling rates of larger particles would also minimise any potential risk for filter feeders. However, finer particle sizes are known to achieve a more complete coverage of sediment by capping agents and to have higher efficiency of P removal, and may therefore be preferred for future applications. Choice of the material and its size must therefore be made with caution, considering potential effects on filter feeding zooplankton; finer particles could lead to adverse food web or water quality effects, such as those observed for some Alum applications (e.g. Schumaker et al., 1993; Paul et al., 2008) and for Phoslock™ (Robb et al., 2003).

Burial of zooplankton eggs was also likely not a problem during the current Z2G1 application. Zooplankton diapausing eggs are less successful at hatching when buried in sediments than when exposed to open water (e.g. Vandekerkhove et al., 2004; Bailey et al., 2005), and burial might therefore prevent a high proportion of emergence. However, even if zooplankton were unable to emerge from below the sediment that was capped, the area of the application was designed to occur only where water overlying sediments becomes anoxic, leaving significant areas that should remain unaffected by the application. The abundances of zooplankton diapausing eggs in the sediments of Lake Okaro are known to be extremely high (Duggan et al., 2002b), which will also ensure high numbers are able to emerge. Visual inspection of the sediment showed, however, that small quantities of the capping agent had settled out in non-targeted areas (i.e. outside of the 0.2 km^2 area). This has implications on the application of Z2G1. Dose rate is likely an important factor influencing the extent of bottom coverage. While the current dose rate of Z2G1 in Lake Okaro (350 g m^{-2}) appeared to have little influence in non-target areas, higher dose rates may bury such areas to a greater extent, potentially reducing hatching rates of zooplankton in the following months. Accurate placement of the material is therefore important to achieve high efficacy, and will also minimise the risk to zooplankton diapausing eggs.

Phytoplankton communities also did not change in composition with the addition of Z2G1 in the time period considered. These species are unlikely to be influenced directly by Z2G1 application, although a long-term change could be expected with reduction in P levels. For example, Lake Okaro underwent a shift in phytoplankton community structure in the early 1960s, from no observed cyanobacteria (Jolly, 1977) to persistent cyanobacteria blooms (Vincent, 1987; Paul et al., 2008). Ideally, the long-term effects of lake restoration at Lake Okaro, including Z2G1 application, would result in a return to pre-eutrophication phytoplankton communities. The most probable cause of any effect on phytoplankton community structure, had it occurred, would have been indirectly through release from zooplankton grazing if these had been affected, as well as from changes in availability of nutrients. In the long term, zooplankton are likely to change in composition with reduction of nutrient and algal levels. Duggan et al. (2002a), for example, found clear patterns in zooplankton community composition associated with trophic state in New Zealand lakes. If Z2G1 is successful in reducing nutrient concentrations in Lake Okaro, then a lower incidence of cyanobacteria can be expected accompanied by a shift in zooplankton species composition.

Conclusions

In this study, we investigated the impact of sediment capping using a modified zeolite, Z2G1, on zooplankton and phytoplankton community structure in Lake Okaro. The addition of Z2G1 to the surface of the lake for the control of internal nutrient loading resulted in no discernable changes in plankton species composition in the water column in the short term, possibly because the sediment capping agent was applied as a coarse-grained material that settled out of the water column rapidly. As P-inactivation agents, including Z2G1, can be milled to finer grain sizes, these finer materials require assessment for their effect on plankton, and a hypolimnetic application is recommended to avoid undesirable direct and indirect effects on plankton communities.

Acknowledgements The first author was funded with a Ph.D. scholarship within the Lake Biodiversity Restoration program funded by the N.Z. Foundation of Research, Science and Technology (Contract UOWX 0505). We thank Britta Deichmann for assistance with phytoplankton identifications and Michael Landman at Scion Research for helpful comments on this article. We gratefully acknowledge Environment Bay of Plenty and Scion for funding assistance. We are grateful to the two anonymous referees whose comments greatly improved the manuscript.

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