The potential for Charophyte re-establishment in large, shallow, eutrophic lakes with special reference to Lake Waikare,

New Zealand

A thesis submitted in partial fulfillment of
the requirements for the degree of
Master of Science in Biology
at
The University of Waikato
by
Aareka Hopkins 2006

KO TE TANGATA

The
University
of Waikato
*Te Whare Wānanga
o Waikato*

Abstract

Lake Waikare is a large, shallow eutrophic lake devoid of submerged macrophytes. I investigated potential methods for re-establishing submerged macrophytes in the lake. Specifically, I subjected charophyte (*Chara corallina*) plantlets to two treatments of exposure in the lake (in areas exposed and sheltered from wind) to test for survival and growth under these conditions, and inside and outside fish exclosures to test for growth and survival in the presence of fish.

While plantlets grew outside the exclosures in winter, their accumulated biomass over 21 days was less than protected plantlets. In winter, the accumulated biomass was lower outside than inside exclosures (by ~40%) at the sheltered site and was lower outside than inside exclosures (by 43%) at the exposed site. Overall, growth rates in winter were higher at the sheltered site (compared to the exposed site) by ~7%. In summer, charophyte accumulated biomass inside the exclosures increased by 85%, while at the sheltered site accumulated biomass increased by 58%. Outside the exclosures in summer the plantlets were completely removed at both sites. Overall, growth rates where higher at the exposed site than the sheltered site by 31%. Fish were responsible for the partial removal of plantlets in winter and total removal of plantlets in summer, and therefore affect the survival and growth of charophytes in Lake Waikare. The embayment at the sheltered site provides the best location in winter for re-establishment of charophytes from oospores because better growth rates were obtained there, and its sheltered location provides protection from severe wave action found at the exposed site. Oospores did not germinate after being submersed in the lake for 90 days due to heavy sedimentation.

To induce an improvement in the present light climate, Alum was tested to determine its effectiveness and longevity for settling lake sediments to allow charophytes to establish and grow. Examining the settling rates of Lake Waikare sediments and water treated with Alum over a range of suspended sediment concentrations and time intervals, sediments settled faster with Alum than without for at least 15 days (at 200 g l^{-1} suspended sediment concentration) and it remained active to 60 days but at reduced effectiveness. At the other concentrations tested (100 g l^{-1} and 300 g l^{-1} suspended sediment concentration), Alum responses were insignificant.

An improved light climate achieved by fish removal or Alum treatment will likely not be sufficient to permit the re-establishment of submerged macrophytes due to the turbid, algal-dominated state of the lake. The present nutrient and sediment levels, wave climate and fish influence must be mitigated so charophyte plantlets can be established.

Acknowledgements

He mihi

Huutia te rito o te harakeke, Kei whea te koomako e koo.
Whakatae rangatira, Rere ki uta, rere ki tai.
Ka patae te patae, He aha te mea nui o teenei ao?
Koo te whakautu,
He tangata, he tangata, he tangata.
Tihei Mauri Ora!

Very special thanks to my supervisors Dr Ian Duggan for his unwavering support and Professor David Hamilton who was always there when I needed him. To Grant Tempero and Wendy Paul I owe a debt of gratitude. To Catherine Beard for advice on working with charophytes and use of field equipment; to Tracy Jones and Wade Tozer in Plant Physiology for their advice and use of equipment; to Mary de Winton at NIWA for advice and help with exclosure design. To Mathew Osborne, Mathew Prentice and Gil Veal I am grateful for their help in the field. To my two trainee technicians grand-nephew Clayton Scott, and Narni Hillman for their assistance in the laboratory and to Technicians Alex Ring and Lee Layboyrie whose help and advice I couldn’t do without.

I make special thanks too to the community of Te Kauwhata, to the Te Kauwhata Community Committee, Brian and Jan White, Jackson Engineering, Carley Transport and Mrs Lexie Cheyne, and to Maurea Marae kaumaatua, in particular Sam and Rosie Tumai and Mrs Lillian Clark for blessing my project.

I am grateful for the financial assistance from Tuapapa Putaiao Maaori Fellowship, Tainui Raupatu Lands Trust, Taharoa IC2 Incorporation, The Maaori Education Foundation, Trust Waikato and the National Wetlands Trust, without whose help this project would not have been possible.

Ngaa mihinui ki a koutou katoa

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Chapter 1

1. General Introduction

Freshwater ecosystems are sensitive to environmental and man-made stressors (Jones, 2001). Man-made stressors include increases in inputs of nutrients such as phosphorus (P) and nitrogen (N) (Dent *et al.,* 2002), and invasive fauna and flora (Richardson and McDowall, 1987). Eutrophication, for example, is an over-enrichment with nutrients (Hosper, 1998); a phenomenon resulting in the deterioration of lake systems world-wide (Cooke *et al.,* 1993; Jeppesen *et al.,* 1998; OECD, 1982). Aquatic systems respond over much shorter time scales to eutrophication than terrestrial systems; for example, populations of the dominant autotrophs living in freshwater systems (e.g. phytoplankton), can reproduce exponentially in hours or days but in terrestrial systems it can take many years for populations to show large scale responses to such stressors (e.g. trees).

1.1: Eutrophication of aquatic systems

Eutrophication in lakes is caused by increased nutrient inputs in runoff or groundwater from the catchment (Dent *et al.,* 2002; Harper, 1992). The increased loading of these nutrients leads to water quality deterioration and significant loss of biodiversity (Beklioglu *et al.,* 2003). Phosphorus, in particular, is generally considered as the limiting nutrient playing a key role in eutrophication (Lewandowski *et al.,* 2003). Eutrophication commonly enhances the abundance of planktonic organisms, is the primary cause of algal blooms (Nakano *et al.,* 2001) and of associated changes in the lake system which contribute to a reduction of light penetration through the water column (Christofor *et al.,* 1994),

Eutrophication has negatively affected submerged macrophyte vegetation in shallow lakes (van den Berg, 1998) and the reduction of the nutrient load can result in an increased coverage of macrophyte vegetation (van den Berg, 1998). Hyper-eutrophic lakes (e.g., Lake Waikare, New Zealand) are characterised by high algal biomass and low mean depth (Wysujack *et al.,* 2002; Schelske *et al.,* 2003). Because these lakes are shallow and have reduced macrophyte abundance they have turbid water due to sediment re-suspension, and phytoplankton growth also causes shading that leads to the loss of macrophytes (Meijer and Hosper, 1997; Schelske *et al.,* 2003).

Cultural eutrophication is a process of anthropogenic origin (Henderson-Sellars and Markland, 1987). Phosphates derived from artificial fertilisers used in agriculture, changes in the catchment from high biomass forests to grasslands, or sediment that can enter lakes, are examples of causes contributing to a gradual increase in nutrient levels, thus encouraging increases in algal biomass (Schwoerbol, 1987; Bright, 2002). Increases in algal biomass promote periodic decreases in dissolved oxygen levels (through increased decomposition), and decreased light penetration (Henderson-Sellers and Markland, 1987). Water quality is compromised, algal blooms occur (Christofer *et al.,* 1994) and a greater abundance of less desirable fish may result (Chapman and Reiss, 1992; Cook *et al.,* 1993). Eutrophication is generally most noticeable in shallow, lentic water bodies (Henderson-Sellars and Markland, 1987; Schwoerbol, 1987).

The effect of eutrophication on inland water bodies is a major reason facilitating the onset of the collapse of macrophyte communities in lowland, shallow lakes globally, including in New Zealand from the 1970's (Davies - Colley *et al.,* 1993). Increased plankton biomass increases light attenuation, and the bottom sediments, previously bound together by the roots of macrophytes, are re-suspended by wave-action resulting in reduced water clarity. The eventual collapse and disappearance of macrophytes in Waikato shallow lakes of New Zealand for example, was caused by increased turbidity that has prevented their re-growth (Davies - Colley *et al.,* 1993; Reeves *et al.,* 2002; Barnes, 2002).

1.2: Alternate Stable States

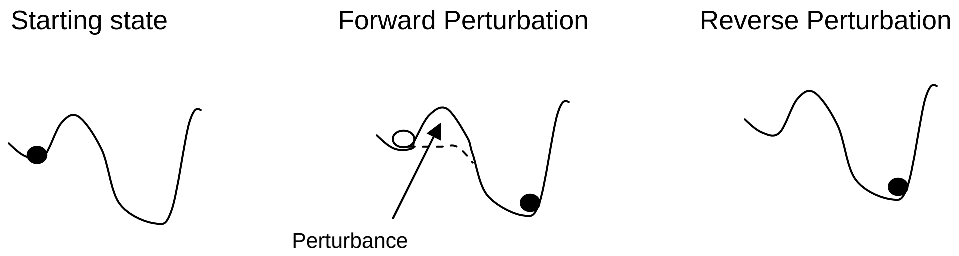
The theory of alternate stable states is a well-known phenomenon (Bachman *et al.,* 1999, 2001; Bayley and Prather, 2003; Blindow, 1997; Dong, 2002). Large shallow lakes assume one of two alternate, stable states: either 1), algal-, or; 2), plant-dominated (Beisner *et al.,* 2003; Blindow, 1993; Hosper, 1998; Moss, 1990), and both states may be extremely stable over time (Scheffer; 1998). Algal-dominated lakes are characterised by poor water quality due to high concentrations of algae (Scheffer, 1998), sediments (Bachmann *et al.,* 2001) and nutrients (Lowe *et al.,* 2001). Plant-dominated states in contrast, are characterised by abundant growths of submerged plants (Bayley and Prather, 2003) and clear water (Scheffer, 1998). Feedback systems centre on the interaction between submerged plants and turbidity. Submerged plants enhance water clarity whereas high turbidity prevents the growth of submerged plants because light is limited by turbidity and submerged plants can only grow to a depth to which light is available i.e. the euphotic zone (Davies-Colley *et al.,* 1993).

There are positive effects of vegetation on water clarity. Vegetation reduces suspended sediments (Hosper, 1997), they provide a refuge for algae grazing zooplankton from fish (Scheffer, 1993), they suppress algal growth by competing for P and N in the water reducing nutrient availability for algae, and release allelopathic substances during their growth phase that are toxic to algae (Sondergaard and Moss, 1997). Lakes with high submerged macrophyte covers have higher transparency than lakes with the same nutrient state in which vegetation is sparse or absent. Vegetation stabilises the clear water state in shallow lakes up to relatively high nutrient loadings. Once the system has switched to a turbid state, a large reduction in nutrient inputs is required to enable plants to re-establish (Scheffer *et al.,* 1993)

Large, shallow lakes exhibiting alternate states can be resistant to vegetation shifts (Bayley and Prather, 2003). The loss or removal of aquatic macrophytes can facilitate a long-lasting switch to algal-dominance and the tendency of aquatic systems to stay in the same state despite changes in external conditions is called hysteresis (Scheffer, 1998). Hysteresis indicates that a system has alternate stable states over a range of conditions (Scheffer and Jeppesen, 1997)

Alternate stable states are restricted to a limited range of environmental conditions (e.g. nutrient inputs) and increases in other conditions (e.g. external nutrient loading) may lead to a stable algae dominated state (Scheffer and Jeppersen, 1997). For example, shallow lakes are prone to constant mixing of bottom sediments by wave action (Van Ness, 2002) but P is released during regular wind and wave events at rates 20 – 30 times greater than that released from undisturbed sediment (Sondergaard *et al.,* 1992). Phytoplankton use this supply of nutrients but the shading effect of phytoplankton (Schelske *et al.,* 2003) can prevent a clear-water phase developing of sufficient duration to allow the establishment and survival of submerged vegetation. Hysteretic responses therefore, limit the ability to restore lakes. The analogy of a ball able to roll forward into the valley once a perturbance has occurred (depicted in Figure 1), illustrates the magnitude of the reverse switch to allow the ball to roll back to its original position i.e. from the lakes present turbid state to a clear water state (Scheffer, 1998). If lakes have elevated levels of nutrients (eutrophic) it is difficult to shift from an algae-dominated state to a plant-dominated state.

Figure 1: Hysteresis resulting from a parameter perturbation causing changes in a lake ecosystem and the equal and opposite perturbation required to move the ball to its original state



1.3: Light Climate in Water

All ecosystems capture light from the sun for their energy (Kirk, 1983). Light contacting the surface of water is subjected to three processes: 1): reflection, 2): scattering, and 3): absorption (Schwoerbel, 1987). The position of the sun dictates the amount of light reflection; the greater the departure of the angle from the perpendicular, the greater the angle of reflection; when the surface of the water is disrupted (e.g. by waves), the angle of reflection increases (Wetzel, 1983). Light scattering is the deflection of quanta by the molecular components of water and the degree of scattering depends on the number of suspended particles in the water (Schwoerbel, 1987) and their concentration (Eisma, 1992). Scattering can vary with water depth, season and location in the lake (Wetzel, 1983) and what light is absorbed, is determined by dissolved inorganic and organic substances in the water (Schwoerbel, 1987). Of the total light entering the water, a portion is scattered and the remainder is absorbed by water, dissolved compounds and suspended particulate matter. The total diminution of radiant energy (by both processes) is called light attenuation (Wetzel, 1983).

The light climate available to macrophytes depends on irradiance attenuation and the diurnal and seasonal cycles of sunlight (Davies and Colley *et al.,* 1993). The maximum depth for macrophyte colonization of lake waters is often determined by light penetration (or irradiance) for photosynthesis to occur (Davies-Colley *et al.,* 1993). This depth is where irradiance (measure of the radiant power per unit area) is reduced to the compensation value i.e. where photosynthesis and respiration are in balance (Schelske *et al.,* 2003); nil net growth occurs at this depth. An approximate index of depth above which Photosynthetically-active Radiation (PAR) is sufficient for aquatic plant growth is the euphotic depth and this depth is taken conventially where PAR (measured as scalar irradiance) has declined to 1% of the surface value (Kirk, 1994).

2: Options for Improving Lake Water Quality

Macrophytes can establish in shallow, light-limited eutrophic lakes with high nutrient levels (P and N) (Blindow, 1992) and charophyte species have many desirable attributes that makes them the target plant-group in lake restoration projects (Dugdale *et al.,* 2005). But fish impact adversely on the turbidity of shallow lakes and methods to address this phenomenon are considered here.

2.1: Charophyte Establishment

Charophytes are highly-developed green macro-algae (Morris *et al*., 2003) that grow in mainly alkaline, freshwater lakes and ponds (van den Berg *et al.,* 1998). They are sensitive to mechanical damage by bottom dwelling fish (Blindow, 1992) and water clarity (Casanova and Brock, 1999). Their growth rates can be affected by more competitive plants (Wade, 1990), variable light and nutrient treatments, water depth, P concentration and substratum particle size. Consequently, they are often absent from highly eutrophic lakes (Forsberg, 1964).

Charophytes are rapid, primary colonisers (Dugdale *et al.,* 2005) and are ideal for rapidly forming a vegetated lake bed. They reproduce vegetatively forming a root attachment to the sediment with colourless, rhizoidal cells (Casanova, 1994) and by inter-seasonal germination of oospores (de Winton, 2004; Casanova, 1994; Haas, 1994). Their low growth-form binds lake sediments (de Winton *et al.,* 2004; van den Burgh, 1998) forming dense mats or “meadows” (Blindow, 1992). Charophytes reduce wind-activated re-suspension of sediments (Korner, 2001; van den Burgh, 1998), act as “refugia” for phytoplanktivorous zooplankton

(Korner, 2001; Coops and Doef, 1996) and provide competition for phytoplankton for P (Blindow, 1992). These are features that enhance water quality and their re-establishment is seen as a desirable outcome in the rehabilitation of eutrophic lakes (de Winton *et al.,* 2004). Charophytes are a native species and were the dominant submerged macrophyte species in New Zealand prior to European arrival (Wood and Mason, 1977).

2.2: Fish Biomanipulation

Biomanipulation can be defined as the deliberate adjustment of biota and their habitats to attain a condition which users (humans) consider more beneficial (Reynolds, 1994). Biomanipulation of fish species has shown positive results when fish were added (Carpenter *et al.,* 1985) (e.g. controlling planktivorous fish species), and when fish were removed (Phillips, 1994). Increasing piscivore biomass decreases planktivore biomass; this increases herbivore biomass (i.e., zooplankton) resulting in decreased phytoplankton biomass (Carpenter *et al.,* 1985; Cooke *et al.,* 1993). Conversely, lowering the biomass of fish that potentially consume or destroy macrophytes (Scheffer *et al.,* 1993) can stimulate macrophyte re-establishment (de Winton *et al.,* 2002; Hosper, 1997; He *et al.,* 1994; Jeppesen *et al.,* 1997)

2.3: Flocculation

Flocculation is an aggregation process occurring in aquatic ecosystems leading to the formation of large particles from small particles suspended in natural (or engineered) water (Leppard and Droppo, 2005). Flocculation is described as "the coalescence of a finely divided precipitate into larger particles (Collocott, 1971) and can regulate the quality of the surrounding water by effecting particle transport down the water column (Droppo *et al.,* 1997) forming large particles from small particles on the way. Their increased diameter causes them to become less dense and settle out more readily (Eisma, 1992) than natural flocculation (Welch and Shrieve, 1994) resulting in improved water clarity, light penetration and primary production (Droppo *et al.,* 1997).

3: Background to Study Site

3.1: Lake Waikare

The Waikato region is the heart of North Island, New Zealand's dairy industry (Fenton, 2002) and is part of an agricultural sector that inflicts the greatest overall impact on fresh water bodies in New Zealand (Bright, 2002). Decreased water flow rates and decreased water levels, including quality and quantity impacts on water in aquifers, streams and lakes receiving agricultural drainage water, are adverse effects emanating from the agricultural sector. Mitigating the inputs of contaminants (e.g. P use in artificial fertilisers) to surface and ground waters from point and non-point sources is the major challenge facing agriculture (Bright, 2002).

Plate 1: Lake Waikare Location and views South west (top) and North west (lower)

Lake Waikare (map reference NZMS 260:S13 040162) is a large (34.4 square kilometers), shallow (1.4 metres) riverine lake located near the North Waikato town of Te Kauwhata. Lake Waikare is the largest lake in the Waikato, it is exposed to the prevailing south-westerly winds and has suffered the effects of eutrophication (Barnes, 2002) since the area was developed for agriculture (McLea, 1986). The effects of erosion, earth works and chemical fertilisers on Lake Waikare have resulted in increased levels of nutrients in the water after 60+ years of agricultural activities (Barnes, 2002). In 1965, the Lower Waikato Waipa Flood Control Scheme was implemented resulting in lowered lake levels, natural lake level fluctuations were discontinued and water inflows were reduced (McLea, 1986). The macrophyte population in the lake collapsed in the 1970's (Davies - Colley *et al.,* 1993; Reeves *et al.,* 2002).

3.2: Previous Studies on Shallow Lakes in Waikato, New Zealand

Several studies on shallow lakes in Waikato, New Zealand have focused on potential remediation strategies to return lakes back to macrophyte dominated states. Stephens *et al.* (2004) conducted a water-drawdown experiment by removing Lake Waikare sediment cores to their research premises to determine if sun-dried sediment cores will remain consolidated once they were re-wetted. de Winton *et al.* (2002) conducted biomanipulation experiments in a small (54 ha), shallow (~2 m), eutrophic Lake Rotoroa (37^{o}48\`S; $175^{o}$16\`E). Between 1994 and 1998, a macrophyte collapse reduced coverage to <1% in the lake. de Winton *et al.* (2002) found charophyte establishment was greater inside exclosure cages compared to outside, suggesting that fish were responsible for the reduced biomass of charophytes exposed to disturbance and grazing. Reeves *et al.* (2002) conducted a study of the environmental changes, wave climate and water quality at Lake Waikare. They outlined historic changes at Lake Waikare and the potential barriers preventing the re-establishment of submerged macrophytes concluded that mitigating the levels of nutrients entering the lake was required for any realistic improvements in water quality to be achieved. They identified a range of potential barriers to submerged macrophyte re-establishment including: the present light climate in the water; high total suspended sediment levels (up to 400 g m^{-3}); high turbidity (>170 NTU's); fish influence (200+ kg ha^{-1} - B. Hicks, University of Waikato, pers. comm.), and; a lack of plant inocula. The ability of lake sediments to sustain vegetation was also questioned and overcoming all or some of these barriers is pivotal to improving the light climate in the lake.

4: Research Objectives

My study examines potential methods for re-establishing submerged macrophytes in Lake Waikare. I investigated whether charophytes could potentially establish and grow in Lake Waikare at current fish densities, or if fish inhibited this establishment and growth. At present no submerged macrophytes inhabit the lake. Further, I investigated the potential of Alum to improve the light climate for potential submerged macrophyte re-establishment by examining the rates of settling over a range of suspended sediment concentrations.

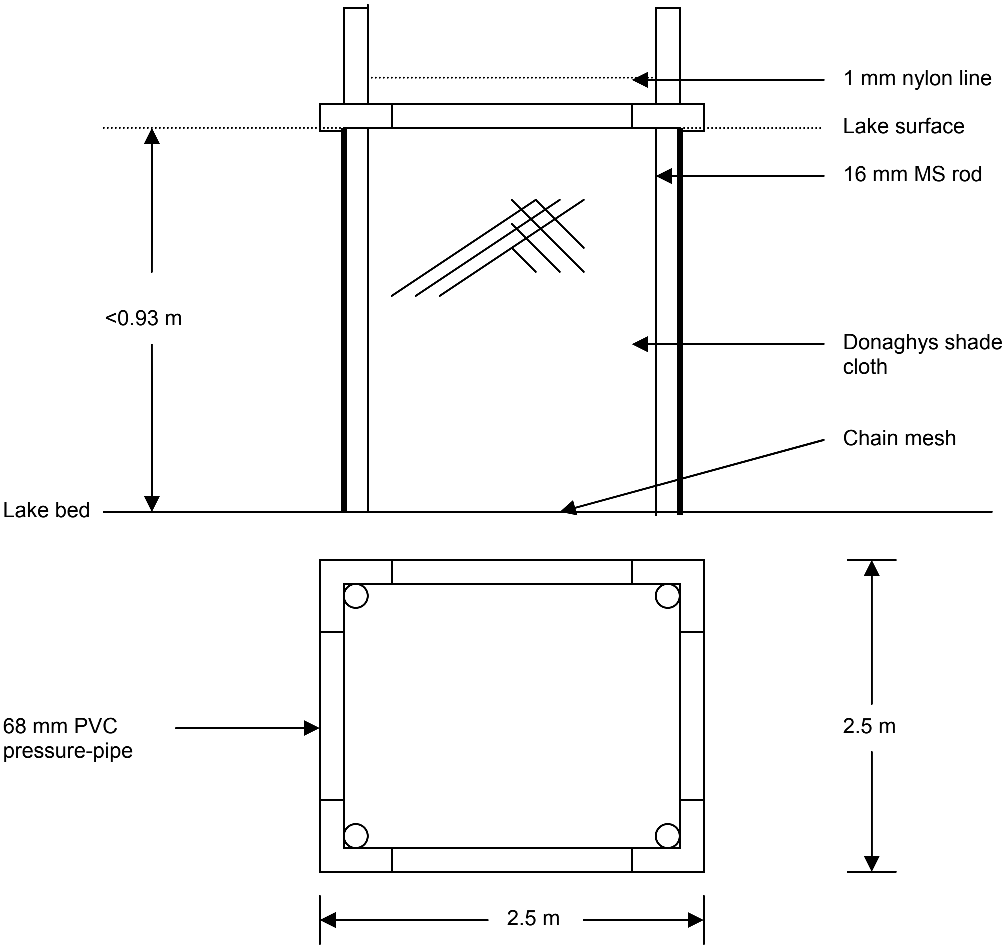
Chapter 2

2. Methodology

2.1.1: Fish Biomanipulation Experiments

Exclosure cages were used to test for the potential establishment and survival of charophytes (*Chara corallina*) in Lake Waikare by excluding fish from potted chara plants placed inside the cages. Plastic protective mesh (shade cloth) formed the cage walls permitting unrestricted circulation of water inside the cages. Exclosure cages were constructed similar in design to those of de Winton *et al.* (2002) in Lake Rotoroa, using four, 2.4 m lengths (68 mm Ø) PVC pressure pipe filled with polystyrene fragments. The pipes were joined in a rectangular configuration (see Figure 2) using elbows permanently fixed in position with PVC jointing compound to form the float system. A 10 m length of Donaghys Hortshade cloth, two metres wide (pore size 7.5 mm $^{-2}$), was used to fashion the cage walls to be suspended ≤2 m below the float system. Along both length-wise edges, a 25 mm hem was sewn and the two unsewn ends where joined (by sewing) to form a caged area of ~2.3 m^{-2} at the lake bed. Four lengths of 15 mm alkathene water pipe (~2.4 m long) were inserted into the hem through cut slots allowing reinforced attachment (using cable ties) of the mesh to the PVC floats. Four millimetre (link diameter) chain was inserted into the hem at the lower end of the mesh to provide anchorage in the lake sediment.

Figure 2: Schematic of Exclosure cage (not to scale)



To secure the cages in position in the lake, two metre length (16 mm Ø) mild steel rods were inserted into the lake bed at each internal corner leaving sufficient height above the cages to prevent dislocation during severe wave motion and to permit transverse and lateral movement of the float. Plate 2 shows the completed cages in storage and *in-situ* in Lake Waikare.

Plate 2: Constructed Cages in Storage and *In-situ*

In storage *In-situ*

2.1.2: Water Quality Parameters

To determine parity of water quality conditions inside and outside cages, a range of water chemical and physical parameters were monitored each 7 days of the 21 day duration of the experiments. Dissolved oxygen (DO), pH and PAR readings were conducted in the field. A portable DO meter (YSI Model 55/12 FT) measured sub-surface oxygen content (%) above the submerged plantlets and a hand-held pH meter (pHScan WP 2) measured pH levels inside and immediately outside the cages. A Licor LI-1400 data-logger light meter measured the quality of light (units: µmoles photons m^{-2} s^{-1}) at sub-surface, mid-column and lake bed levels.

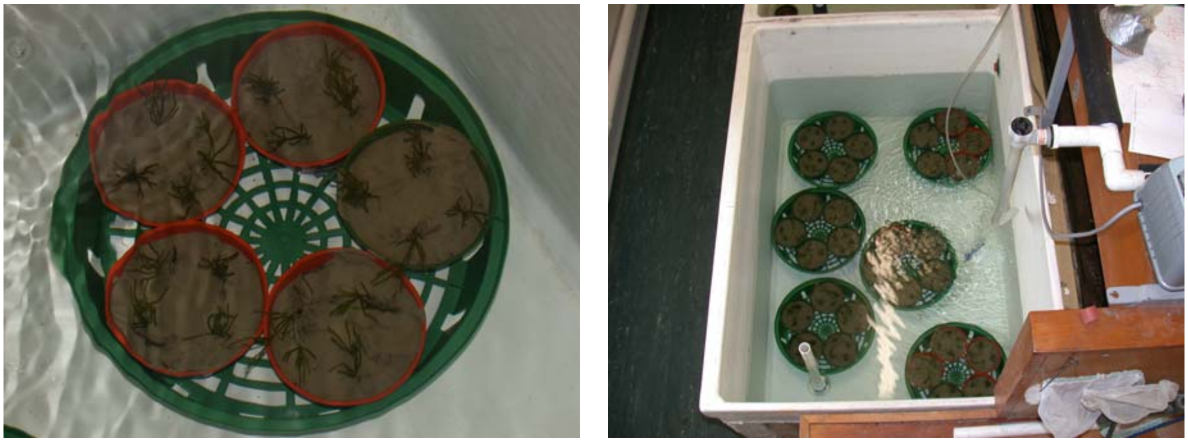
Duplicate water samples (~100 mls) were collected from inside and immediately outside each cage at the start, and at each seven-day interval of the experiments. The samples were transported to the laboratory on ice in a chilly bin and known volumes of water were immediately filtered in dim light for chlorophyll *a* samples. These filters were wrapped in aluminum foil, and frozen for later analysis. Suspended-sediment concentrations were determined by filtering a known volume of water (10 mls) through identified, pre-weighed filters and re-dried to a constant weight in an oven at 103°C for three hours. A DROTT turbidity meter was used to measure water clarity (units: Nephelometer turbidity units) in the laboratory.

2.1.3: Charophyte Collection, Establishment and Deployment

Lake Waikare sediment, and charophyte plantlets collected from Lake Rotoroa were used for biomanipulation experiments in Lake Waikare. Sixty kilograms of sediment was collected with an Ekman grab (vol. ~6×10^{-4} m^{-3}) cast from a boat on the 21st July, 2005 (water depth <0.93 m), transported to the laboratory and stored in a refrigerator at 10^{o}C. The sediment was homogenised in the laboratory by stirring and mixing thoroughly and 300 gram sub-samples (50 mm depth) were decanted into five gardening pots (surface area 5.67 mm^2) which were secured in a plastic holding-dish to facilitate ease of deployment in and recovery from the water.

Chara corallina plantlets were collected from Lake Rotoroa on the 20th July, 2005 using a long-handled, garden weeder (water depth ~2m) to scoop them into a boat. In the laboratory, apical stems containing three nodes were removed and five plantlets were carefully inserted into each of the previously prepared pots of sediment. The plantlets were inserted to a depth sufficient to completely bury the lower-most node creating a dish of potentially 25 viable charophyte plantlets (Plate 3).

Plate 3: Chara Plantlets undergoing rhizoidal root attachment in the Laboratory



Prepared Dishes of Chara Plantlets

Plantlets immersed in De-chlorinated water

Sixteen dishes were prepared in this way and a further 25 pots (125 plantlets) were prepared to derive a starting biomass dry-weight (DW). Plants were submerged in a tub (Plate 3) to a depth of ~ 0.5 metres (m) in de-chlorinated water for five weeks to facilitate rhizoidal root attachment. Dechlorinated water is achieved by passing tap water (containing chlorine) through a bed of activated carbon that removes certain contaminants (added chlorine) without affecting the ionic composition of the water (N. Ling, Waikato University, pers. comm.). Water was added to the tubs at a constant rate of 15 l hr^{-1} to permit water movement across the plantlets. A 16 hour light regime was employed at 15 micromoles m^{-2} sec^{-1} light intensity.

On 20 September, 2005, the dishes of plantlets were fitted with Allflex, permanently-marked, bovine ear tags for secure identification and transported to the experiment site at Lake Waikare. At the lake-shore, sediment traps were fitted, buoy lines were attached and re-enforcing mesh (600 x 200 mm - mesh size 50 mm) was fixed to the dishes with cable ties to provide stability at the lake bed. The dishes were transported (by boat) to the exclosure cages (deployed the previous day) and lowered carefully inside and outside the cages ensuring their locations were carefully recorded.

Reeves *et al.* (2002) in their study of Lake Waikare stated macrophyte seed banks were virtually non-existent, possibly due to burial by sporadic, heavy suspended-sediment loading (>400 g m^{-3}). The summer plantlet experiment mirrored the winter experiment but was conducted in conjunction with the oospore experiment, i.e. the plantlets were deployed in the same cages as the oospores on the 22 December 2005 to test their survival and establishment.

Chara species' oospore-laden sediment was collected on the 3 December, 2005 with an Ekman grab cast from a boat (water depth 2 m) from Scooter Bay (Lake Rotoroa). This sediment was added to sediment previously collected for *Chara* species identification, homogenized and assessed for total oospore concentration (see Appendix 1). Seventeen trays of pots (surface area 5.67 mm^2) were prepared in similar fashion to the plantlets (sediment depth 50 mm), 16 were deployed *in-situ* on the 8 December and one tray was placed in an incubator in the laboratory (18^0C and 16:8 light:dark regime) to determine whether germination would occur in optimal conditions compared to conditions in Lake Waikare.

2.1.4: Retrieval of Deployed Charophyte Plantlets and Oospores

Three weeks (21 days) post-deployment of the plantlets, they were retrieved from the exclosure cages and transported back to the laboratory following the collection of final field data and water samples. In the laboratory, the float assembly and steel mesh were removed and the sediment in each pot contained within the 16 individual dishes (n=80) was removed and sieved through a 250 μm sieve to remove the charophyte plantlets in their entirety (shoots and roots). The plantlets were transferred to a labeled pie dish and dried at 80^oC to a constant weight. This method was repeated for all pots. The exclosure cages were removed and stored at the site following the removal of the plantlets.

The summer plantlets were retrieved after 21 days on the 12 January, 2006. Field data and water samples were collected and these were transported on ice (in darkness) to the fish laboratory where the float assembly and steel mesh were removed. The pots of plantlets were sieved and all charophyte material was transferred to a pie dish, labeled and dried (80°C) to a constant weight.

The oospores were retrieved on the 8 March 2006 and returned to the laboratory after 90 days submerged in Lake Waikare. In the laboratory, the float assembly and mesh were removed and sediment from each pot was sub-sampled (n=3), sieved and observed under a microscope to determine whether oospore germination had occurred. The tray of oospores placed in the incubator on day one of the experiment, was removed and sieved (250 μm) for germinated plantlets in similar fashion to the winter plantlet experiment. All charophyte material was transferred to a pie dish, labeled and dried to a constant weight.

2.1.5: Wave Climate

Dobie Wave meters were deployed on the lake bed (way points S37° 24’57.5; E175° 9’47.2, and; S37° 24’58.6; E175°9’37.7) to compare the effects of wind-induced wave pressure at each site. These were deployed when the exclosure cages were deployed.

2.1.6: Aquafauna

Bird fauna resident at the site included Grey and Mallard duck, Shag, Canadian Geese and Black swan. Although web-footed, ducks and shag especially can perch on the floats potentially contaminating the water inside the exclosures. To combat this threat, a 1 mm nylon line was erected 100 mm above the floats supported by brackets at each corner to act as a hazard and deterrent to the birds to perch. Fish fauna resident at the site include: Rudd (*Scardinius erythropthalmus*); Perch (*Perca fluviatilis*); Koi carp (*Cyprinus carpio*); Brown bullhead catfish (*Ameiurus nebulosus*); Banded Kokopu (*Galaxias fasciatus*); Shortfin eel (Anguilla australis) and Common Bully (*Gobiomorphus cotidianus*) (Gary Watson, eel fisherman, pers comm.)

2.2: Flocculation with Alum

Alum (Aluminium Sulphate) was used to flocculate filtered Lake Waikare water prepared with a range of suspended-sediment concentrations to determine if the resultant solutions would settle out faster with additions of alum than without alum. Replicates of prepared lake water were mixed to three suspended-sediment concentrations based on a range of naturally occurring concentrations identified by Reeves *et al.* (2002), and settling rates of replicates treated with alum were compared in settling experiments with replicates without alum. Settling chambers (2) were used to determine if any significant differences in sediment settling rate, pH, light penetration or turbidity could be detected between Alum and no Alum experiments over time. To test the longevity of alum treatment, the experiment was replicated over four time intervals of 15 days.

Reeves *et al.* (2002) showed that mean total suspended solids (TSS) concentration in Lake Waikare for the period October 1992 to October 2000 ranged between 25 g m^{-3} and 400 g m^{-3} and this data prompted the choice of 100 g m^{-3}, 200 g m^{-3} and 300 g m^{-3} as the suspended-sediment concentrations for this study. My experiments were conducted in the laboratory (walk-in refrigerator) at 10^{o} C and the range of TSS reported by Reeves *et al.* (2002) reflects a lake with a large fetch, exposed to prevailing winds able to generate high wave pressure very rapidly resulting in variable suspended-sediment concentrations. Increased settling rates of suspended sediments treated with alum are expected (Welch and Schrieve, 1994) and my experiments intend to confirm this expectation as well as investigate the longevity of a single alum treatment.

High quality water was required to prepare the alum solution and to rinse the interiors of the settling chambers (and syringe) between replicates. Reverse osmosis (RO) is the process of forcing a solvent (tap water) from a region of high solute (contaminants) concentration through a membrane to a region of low solute concentration by applying a pressure in excess of osmotic pressure. The membrane is semi-permeable meaning it allows the passage of tap water but not of solute. The membranes have no pores and the separation takes place in a dense polymer layer of microscopic thickness designed to only allow water to pass through. The water liquefies in the polymer and crosses it by diffusion in a process requiring high pressure to be exerted on the high concentration side of the membrane (usually 5 MPa – 20 MPa/50 bar – 200 bar).

2.2.1: Materials and Methods

Two perspex settling chambers (volume ~11 litres) were fitted with Licor-L1400 Data-Logger light-sensors positioned ~20 mm from the bottom of the chambers (to accommodate wiring). The exterior of the chambers were covered with tin-foil to prevent incidental light entering (or escaping) from the cylinders to ensure data integrity. Lighting was provided by an Ever-poise lamp (75 watt intensity) complete with a shade assembly that fitted snugly over the opening of the cylinders. A 20 ml adjustable, automatic syringe especially adapted with a weighted, clear plastic tube and brass “tee” nozzle was used to collect 15 ml sub-samples of water/sediment solution from the chambers. The chambers were positioned on a work-bench in the refrigerator (see Plate 4).

Plate 4: Settling Chambers and Associated Equipment

2.2.2: Water Preparation

Lake Waikare water (260 litres) was collected and transported to the laboratory. The water was left to settle for 48 hours in the refrigerator to allow the settling out of aggregates and larger sediment particles. The supernatant was decanted and transferred to new, previously rinsed (with RO water), sealable plastic containers (volume ~11 litres) and stored in the refrigerator. Following decanting, the water was filtered through a Whatmans 41 filter circle measured (with measuring cylinder) into 10 litre replicates, re-sealed in their containers and returned to the refrigerator. Twenty four replicates were prepared in this way.

2.2.3: Sediment Collection, Preparation and Concentration Determination

Lake Waikare sediment (10 kg) was collected with an Ekman grab in similar fashion to the biomanipulation experiment, transported to the laboratory and stored in the refrigerator. Replicate sub-samples (n = 5) of wet sediment (~20 g) were dried (80^{o}C for 24 hours) to a constant weight and from these sub-samples a wet sediment weight was calculated to achieve the 100 g m^{-3}, 200 g m^{-3} and

300 g m^{-3} concentrations required of the study. Wet sediment was weighed (9.974 g) (see Appendix 11) and added to previously prepared containers of filtered and measured (10 l) lake water and homogenized. Eight replicates were prepared in this way to provide four treatment replicates and four control replicates with a suspended-sediment concentration of 300 g m^{-3}. The containers were re-sealed, labeled by date, replicate and concentration, and returned to the refrigerator. Replicates for the remaining sediment concentrations were prepared similarly, resealed and labeled accordingly and returned to the refrigerator.

2.2.4: Alum Concentration Determination

A stock solution of alum was prepared using a recipe based on Standard Methods (Clesceri and Greenberg, 1989). Cooke *et al.* (1993) recommended a stock solution (1.25 mg Al ml^{-1} made by dissolving 15.4211 g in distilled water diluted to 1.01. One ml of this stock was added to 500 mls water while monitoring the pH. Cooke and Kennedy (1981) state dose determination should be small enough to prevent drastic shifts in pH – the higher the dose rate, the more pH and alkalinity decrease. The maximum dose is that which exceeds 50 ug l^{-1}, a concentration found to be safe by Everhart and Freeman (1973) for trout.

The final Alum dose rate added to the prepared lake water samples was from a stock solution of 3.2115 g of Alum mixed with 500 mls RO water. Replicates of lake water were dosed with increments of 0.01 mls Alum until pH 6 was reached (Rydin and Shrieve, 1998). Toxicity is not regarded as a problem so long as pH is controlled (Cooke and Kennedy, 1981) and maintained in the 6 – 8 range (Cooke *et al.,* 1993). Al solubility is pH-dependant with maximum floc formation occurring at ~pH 6 – 8 (Cooke *et al.,* 1981). An Alum dose of 5.5 ml l^{-1} was added to the prepared 10 litre lake samples.

The use of Al is a widely applied lake restoration measure for the precipitation and inactivation of P (Lewandowski *et al.,* 2003). Its purpose was to initially lower P content by precipitation and control the sediment P release and inactivation

(Cooke *et al.,* 1993). Alum treatment of polymictic lakes has been successful in reducing the photic zone P concentration and longevity of the treatment has been in the 4 – 9 year range (Cooke *et al.,* 1993). Alum was used in this study to flocculate Lake Waikare water over a range of suspended sediment concentrations in the laboratory.

2.2.5: Settling Experiment Process

The transfer of prepared water samples to the settling chambers (containing aerators positioned at the bottom and mid-point of the chambers), signaled the start (zero time) of the experiments. The aerators ensured solution homogenization of the samples for the measurement of water temperature and pH levels following which they were removed. The Ever-poise light and shade system was positioned securely over the chamber opening and turned on for the collection of PAR (micromoles m^{-2} sec^{-1}) data. An automatic syringe apparatus was used to draw off water sub-samples (~15 mls) from sub-surface and mid-chamber levels as well as at the chamber bottom. These sub-samples were transferred to individual Vulcan tubes (volume 50 ml), labeled carefully and stored in batches in individual plastic bags. The collection of PAR data and water sampling was repeated each 30 mins for the duration of the three hour experiment and at its termination, pH and temperature data was again recorded. The water in the settling chambers was then transferred back to its original container where it was re-sealed, date labeled and stored. Twenty four replicates were tested in random order (see Appendix 111).

At the conclusion of the each batch of settling experiments (zero interval), the stored water sub-samples were analysed for clarity (DROTT turbidity meter) and suspended-sediment concentration. The suspended-sediment concentrations (g m^{-3}) from each level of the settling chamber were required to monitor the settling characteristics of the various replicate sample concentrations over time. Suspended sediment determination was carried out in similar fashion to the biomanipulation experiment and to retain water volumes in each replicate, water sub-samples collected for turbidity analyses were returned to their original containers. The experiments were repeated for a further three, 15 day intervals (to Day 60). At each interval, the containers were re-suspended (by shaking vigorously for 20 seconds) and re-added to the settling chambers where they were again monitored over time as above.

2.2.6: Statistical Methods

Mann Whitney *U* tests (Zar, 1996) were used to test for differences at time 180 minutes in suspended sediments, turbidity, PAR and pH levels.

This was due to differences in sample sizes between Alum treatments (five replicates at 300 g l^{-1} suspended sediment (SS) concentration, three replicates at 200 g l^{-1} SS concentration, and four replicates at 100 g l^{-1} SS concentration, and no Alum treatments (three replicates at 300 g l^{-1} SS concentration, five replicates at200 g l^{-1} SS concentration and four replicates at 100 g l^{-1} SS concentration).

Chapter 3

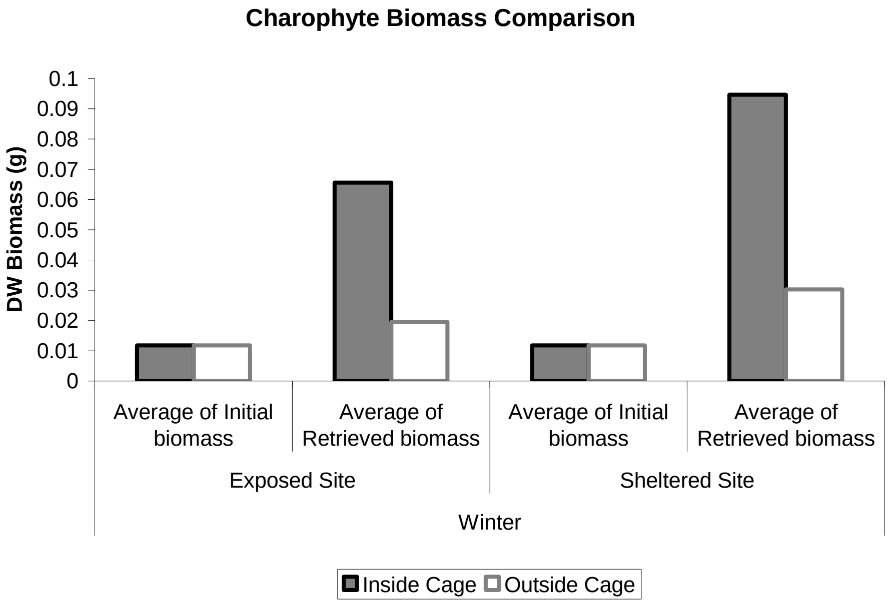
3. Results of Charophyte Establisment Experiment

3.1: Charophyte Plantlet and Oospore Performance

3.1.1: Winter

Mean plantlet dry weight biomass inside the exclosures at the exposed site had increased by 0.054 g from an initial biomass of 0.011 g in winter (Figure 1) an increase of 80%. Similarly, mean plantlet dry weight biomass inside the exclosures at the sheltered site had increased by 0.084 g from an initial dry weight biomass of 0.011 g, an increase of 87%. Outside the exclosures at both the exposed and sheltered sites, dry weight biomass had increased by 0.012 g (45%) and 0.019 g (52%) respectively. Overall, the increase in mean dry weight biomass at the sheltered site was superior to mean dry weight biomass inside and outside the exclosures at the exposed site (by 7%).

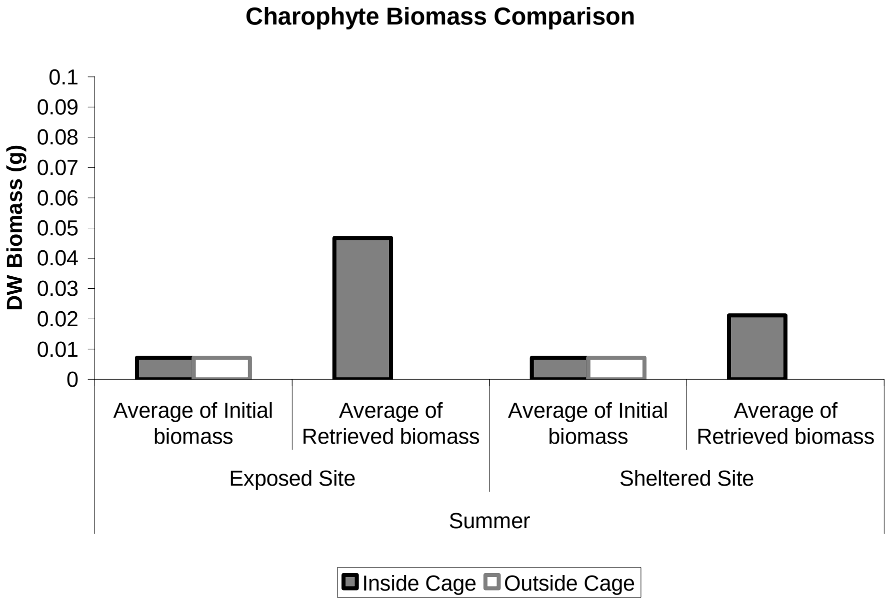
Figure 1: Mean charophyte biomass inside exclosures compared with outside exclosures between exposed and sheltered sites in winter (2005) after 21 days



3.1.2: Summer

At termination of the experiment, summer mean plantlet dry weight biomass inside the exclosures at the exposed site increased by 0.040 g from an initial biomass of 0.007 g (Figure 2) an increase of 85%. Similarly, at the sheltered site mean plantlet biomass had increased ~ 0.010 g inside the exclosures from an initial biomass of 0.007 g (59%). However, those plantlets deployed outside the exclosures at both sites were completely removed or lost. Mean dry weight biomass was ~ 0.030 g (31%) higher at the sheltered site.

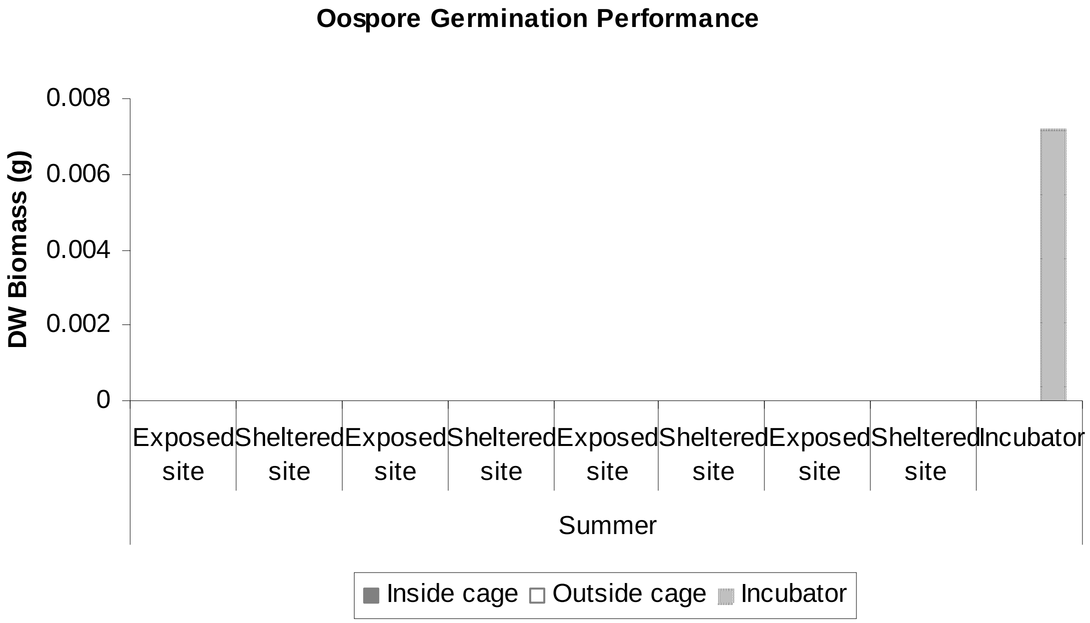
Figure 2: Mean Charophyte Biomass inside exclosures compared with outside exclosures between exposed and sheltered sites in summer (2006) after 21 days



3.2: Charophyte Oospore Germination

No obvious sign of oospore germination was evident on retrieval of the oospores from the exclosures after 90 days in Lake Waikare, but germination was evident in sub-samples incubated in the laboratory. Microscopic inspection of sub-samples (n=3) from each of the retrieved dishes (n=10) confirmed no germination or root development had occurred. The biomass shown in Figure 3 reflects the weight of oospores sieved and inspected compared to the dry weight of plants after 90 days in the incubator.

Figure 3: Mean Oospore germination inside and outside exclosures at the exposed and sheltered sites after 90 days in summer (2006) compared with controlled germination (90 days) in the laboratory

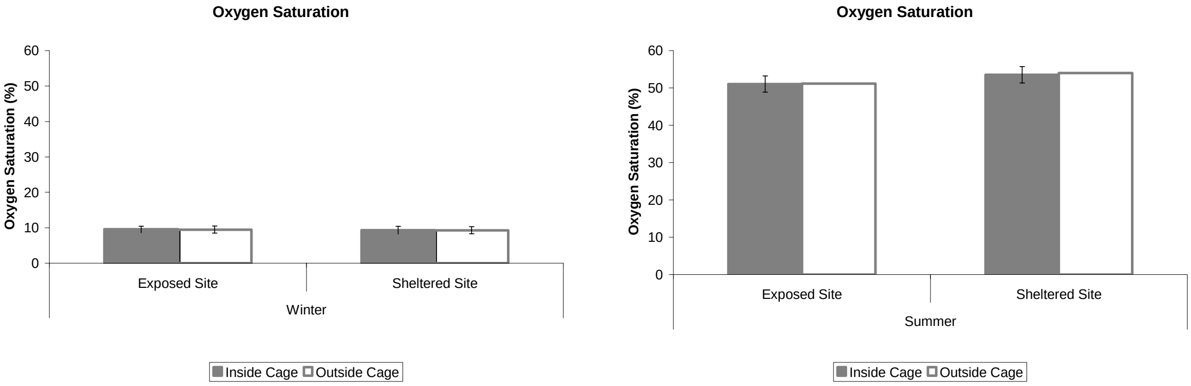


3.3: Physical and Chemical Parameters

3.3.1: Dissolved Oxygen

Mean oxygen saturation (%) was similar between the sheltered and exposed sites in both winter (Figure 4 a) and summer (Figure 4 b) but were higher (by 80%) in summer compared to winter.

Figure 4: Mean oxygen saturation (%) levels inside exclosures compared with outside exclosures between two sites in winter (2005) and summer (2006)



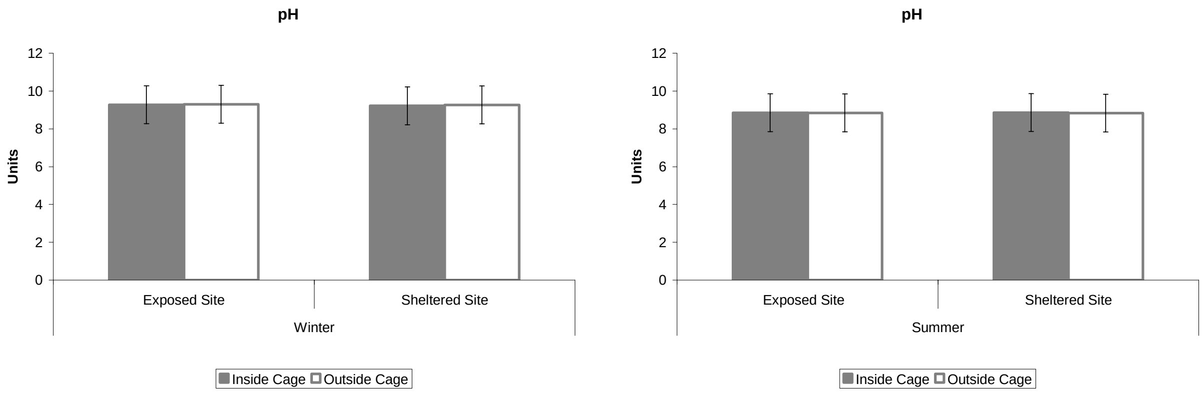
(a) Winter 2005

(b) Summer 2006

3.3.2: pH

Mean pH levels were similar inside and outside the exclosures at the exposed site and the sheltered site in winter (Figure 5a) and summer (Figure 5 b)

Figure 5: Mean pH levels inside exclosures compared with outside exclosures between two sites in winter (2005) and summer (2006)



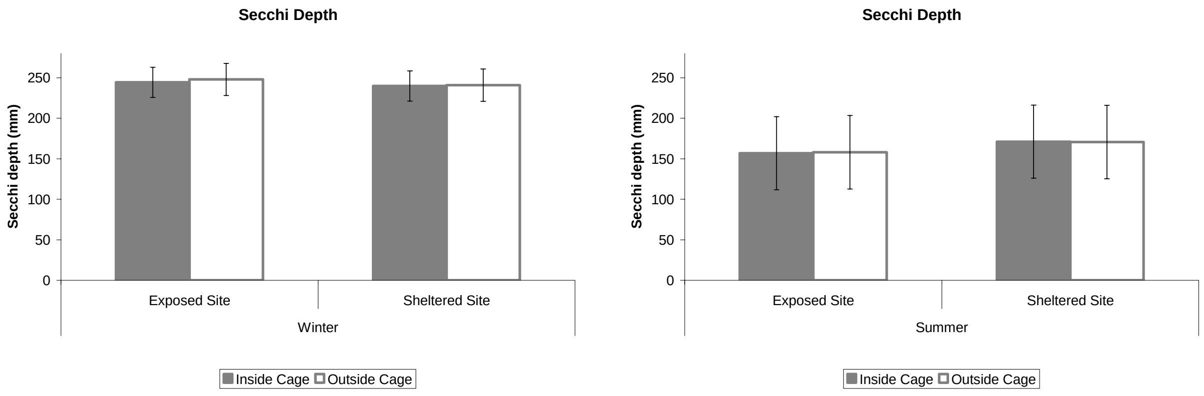
(a) Winter 2005

(b) Summer 2006

3.3.3: Secchi Depth

Mean Secchi depth levels were similar inside and outside the exclosures at the exposed and sheltered sites in winter (Figure 6 a). Mean Secchi depth levels in summer (Figure 6 b) were similar inside and outside the exclosures at both the exposed and sheltered sites, but visibility was better by ~90 mm in winter than summer at both sites.

Figure 6: Mean Secchi Disc levels inside exclosures compared with outside exclosures between two sites in winter (2005) and summer (2006)



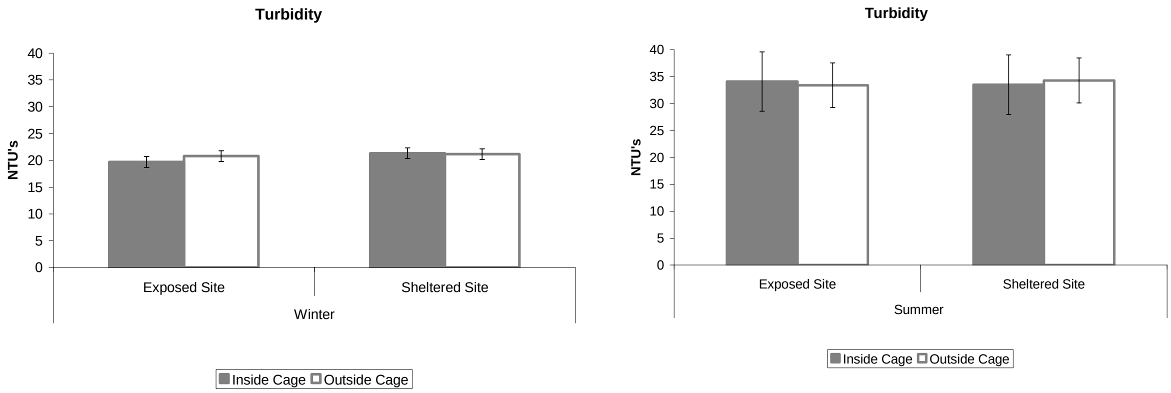
(a) Winter 2005

(b) Summer 2006

3.3.4: Turbidity

Mean turbidity (nephelometer turbidity units, NTU) levels inside and outside the exclosures at the exposed site in winter (Figure 7 a) were similar to mean turbidity inside and outside the exclosures at the sheltered site. Mean turbidity between the seasons was higher by ~14 NTU's ($p < 0.05$) in summer than in winter (Figure 7 b).

Figure 7: Mean Turbidity levels inside exclosures compared with outside exclosures between two sites in winter (2005) and summer (2006)



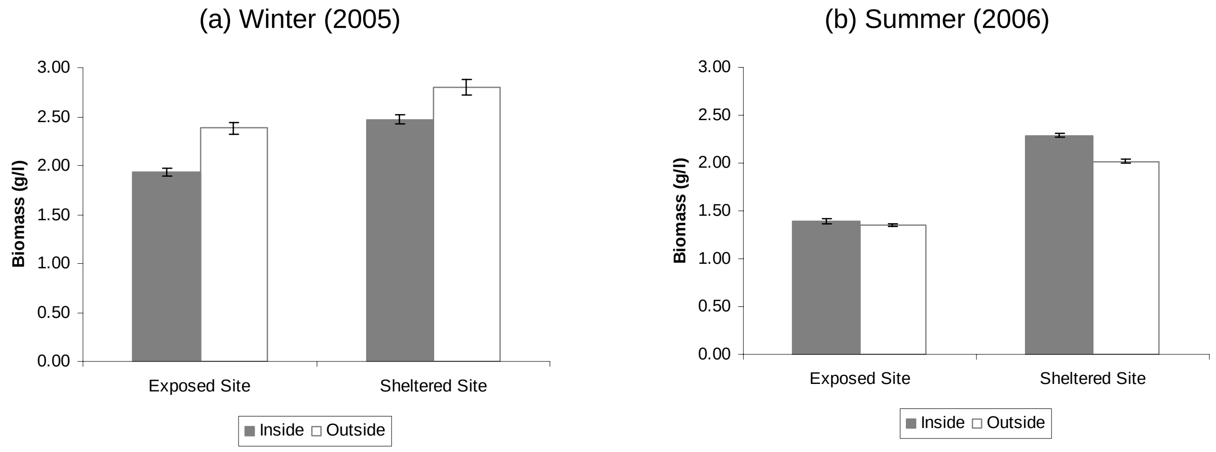
(a) Winter

(b) Summer

3.3.5: Suspended Sediment Concentrations

Mean suspended sediment (SS) concentration outside the exclosures was significantly higher ($p < 0.05$) by 0.500 g l^{-1} (21%) than inside the exclosures at the exposed site in winter (Figure 8 a). At the sheltered site mean SS concentration outside the exclosures was significantly higher ($p < 0.050$) by 0.400 g l^{-1} (9%) than inside the exclosures. Mean SS concentration inside and outside the exclosures at the exposed site in summer (Figure 8 b) was similar, but at the sheltered site mean SS concentration was higher (by 0.300 g l^{-1}) (13%) inside the exclosures than outside. Mean SS concentration was higher (by 8%) in winter than summer.

Figure 8: Mean suspended sediment concentration inside exclosures compared with outside exclosures between two sites in winter (2005) and summer (2006)



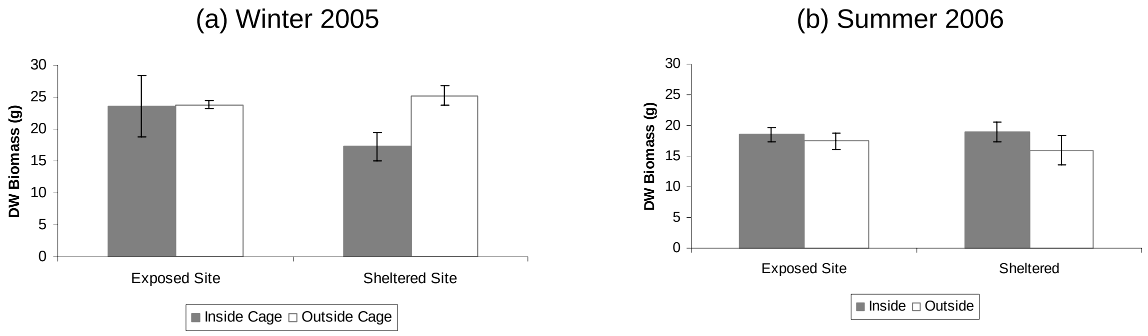
(a) Winter (2005)

(b) Summer (2006)

3.3.6: Sedimentation

Mean sediment dry weight biomass collected from traps inside the exclosures was similar after 21 days, to outside the exclosures at the exposed site in winter (Figure 9 a). At the sheltered site, mean sedimentation was lower by 0.500 g (35%) inside the exclosures than outside the exclosures. Mean dry weight biomass at the exposed site was higher by 1 g (5%) inside than outside exclosures and ~2 g (11%) at the sheltered site in summer (Figure 9 b).

Figure 9: Mean Sedimentation levels inside exclosures compared with outside exclosures between two sites in winter (2005) and summer (2006)



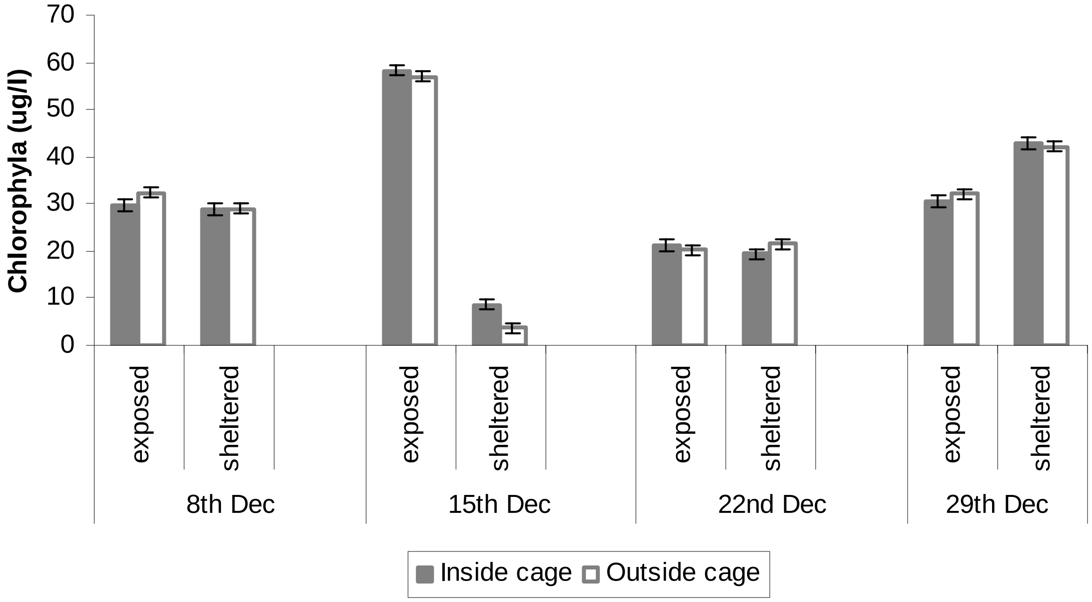
(a) Winter 2005

(b) Summer 2006

3.3.7: Chlorophyll *a*

Mean chlorophyll *a* levels inside the exclosures were generally uniform when compared to outside the exclosures in summer (Figure 10). Mean chlorophyll *a* levels were uniform between the exposed site and the sheltered site on each day except for 15 December when levels were significantly higher at the exposed site

Figure 10: Mean chlorophyll *a* levels inside exclosures compared with outside exclosures between two sites in summer (2006)



3.3.8: Dobie Wave Meters

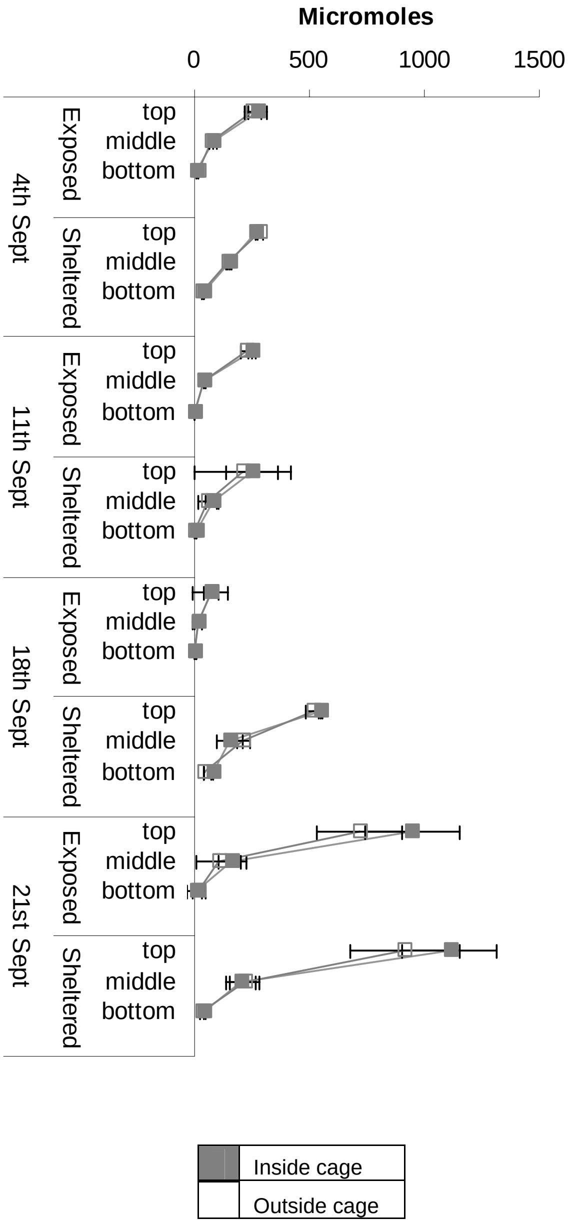
Dobie wave meters monitored differences in wave pressure at the two study sites. Mann-Whitney *U* test results showed there was a significant difference in wave pressure ($p<0.05$) at the exposed site compared to the sheltered site in winter and summer.

3.3.9: Light Climate

3.3.9.1: Winter

Mean light levels inside the exclosures and outside the exclosures at both the exposed site and the sheltered site in winter (Figure 11) were similar with mean light levels inside the exclosures on 21 September ~ 200 micromoles (µm) higher than outside the exclosures at the exposed site and ~ 150 µm higher inside the exclosures than outside the exclosures at the sheltered site. The calculated euphotic depth was similar inside and outside the cages (see Appendix 1V) but higher at the sheltered site (average 1.48 m) than at the exposed site (average 1.15m).

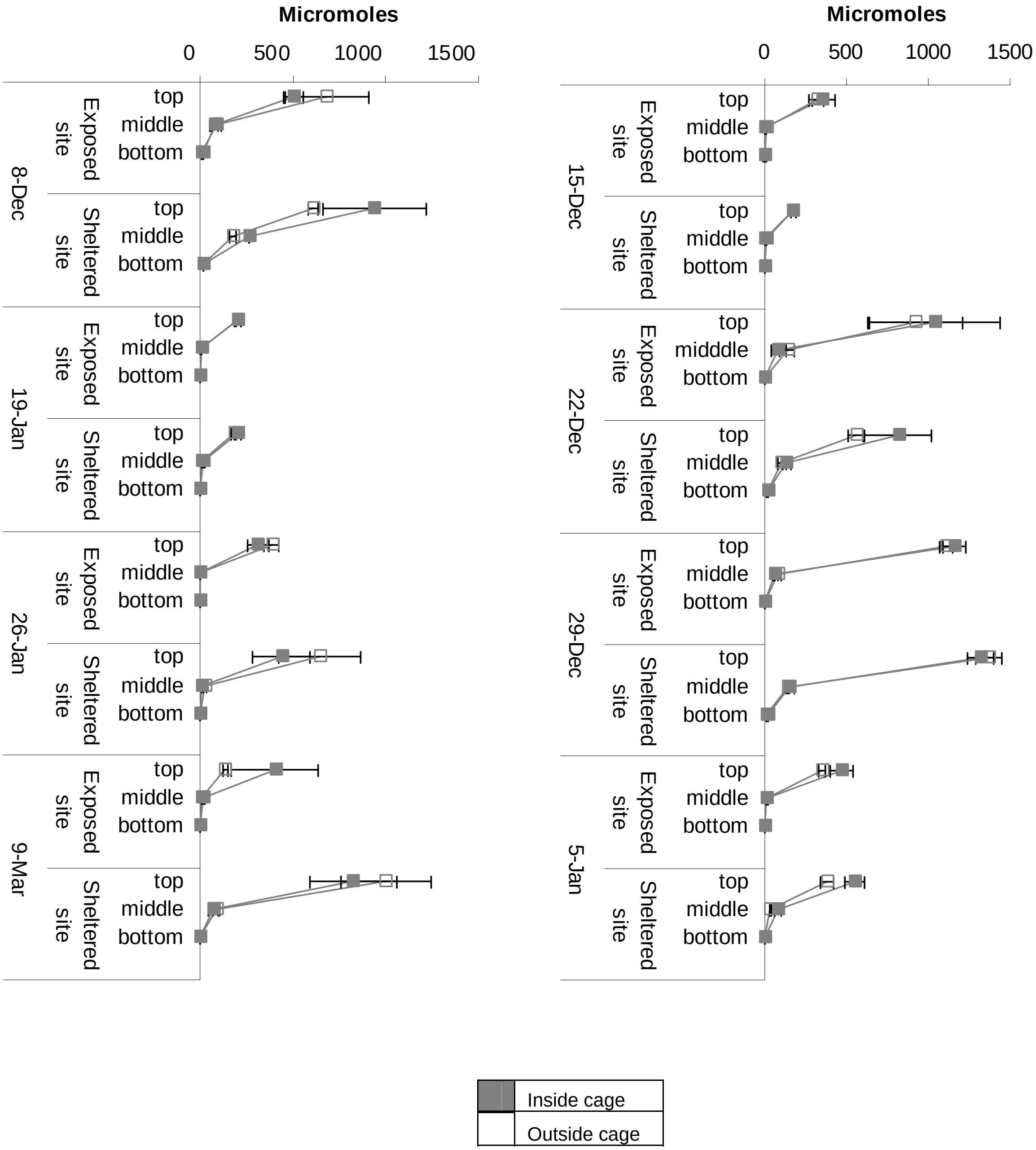
Figure 11: Mean PAR levels inside exclosures compared with outside exclosures between the exposed and sheltered sites over 21 days in winter (2005)



3.3.9.2: Summer

Mean light levels were 100 µm higher inside the cages on 8 December than outside the exclosures at the exposed site in summer (Figure 13 a) when the oospores were deployed but were ~275 µm lower inside the exclosures than outside the exclosures at the sheltered site. Further down the water column, mean light levels were uniform at ~ 50 µm inside and outside the exclosures at the two sites. Mean summer light levels on 15 December (Figure 13 b) when the charophytes were deployed, were ~ 550 µm higher inside the exclosures at the exposed site than outside the exclosures but at the sheltered site, mean light levels were uniform inside and outside the exclosures although lower than levels at the sheltered site. Mean light levels remained uniform inside and outside the exclosures but variable at sub-surface levels. For the duration of the oospore experiment (Figure, 13 a), mean light levels showed signs of variability at sub-surface levels but uniform inside and outside exclosures. The euphotic depth was similar inside and outside the exclosures at both sites and higher at the sheltered site (0.78 m) than the exposed site (0.6 m) (see Appendix 1V).

Figure 12: Mean PAR levels inside exclosures compared with outside exclosures between the exposed and sheltered sites over time in summer (2006) for *Chara* oospores (a) and plantlets (b)



(a) Oospore Experiment Duration

(b) Plantlet Experiment Duration

3.4: Discussion

The growth and survival of charophyte plantlets deployed in Lake Waikare was inhibited in the presence of fish. Winter dry weight biomass of charophyte plantlets protected by exclosures had increased 87% compared to 52% outside the exclosures and variations in the growth rates were caused by either: differences in prevailing conditions (inside and outside exclosures) and between the two sites (exposed and sheltered); seasonal variations such as wave climate, imposed lake-level fluctuations, and fish motility and plant palatability that can affect fish predation in winter. Physical and chemical monitoring results confirmed conditions inside and outside the exclosures were similar except for wave climate, suspended sediments and sedimentation, especially in winter.

The wave climate was significantly greater at the exposed site ($p < 0.05$) in winter and summer. A reduced wave climate at the sheltered site may have been a factor in the sedimentation that occurred to cause reduced plantlet growth outside the exclosures in winter. Calm conditions can promote the settling of suspended particles (Sondergaard and Moss, 1997) and higher suspended sediment levels (by 37%) and sedimentation (by 32%) were recorded here than at the exposed site (Figure 8 b). A water level fluctuation regime operates at Lake Waikare (Moturiki datum range 5.4 – 5.75) that can affect some charophyte species by altering their morphology, particularly with increased water levels (Casanova, 1994) while some species increased in biomass with reduced water levels (Casanova and Brock, 1999). Winter conditions can affect fish metabolism which slows, lowering the requirement for nutrients meaning the plantlets may have been selectively and/or moderately grazed. Submerged macrophytes release allelopathic substances during their growth phase (Faafeng and Mjelde 1997) and this characteristic may have affected charophyte palatability. By contrast, fish are more voracious in summer accounting for the complete loss of charophytes from outside the exclosures at both sites in summer, The fish population in Lake Waikare can inflict direct damage on submerged macrophytes. Fish found in Lake Waikare such as Rudd, Perch, Koi carp and Catfish are capable of causing damage through their foraging action (Jeppesen *et al.,* 1997) or by grazing (Cooke *et al.,* 1993). Rudd are herbivorous (Lake *et al.,* 2002) while Perch and Catfish are benthic feeders. Koi Carp are destructive of aquatic plants directly through grazing and uprooting (McQueen, 1990). Common Bullies were discovered in the sediment on retrieval of three pots from outside the exclosures and small “hollows” were obvious in the sediment in two pots, a characteristic of fish disturbance identified by de Winton *et al.* (2000). Furthermore, the feeding behaviour of fish promotes turbidity of the overlying water thus hampering plant growth (Hosper, 1997). Turbid water is not considered a suitable habitat for charophytes (Casanova and Brock, 1999) and are known to decline under such conditions (Forsberg, 1965) but Casanova and Brock (1999) found they grew in turbid water as I did. Charophytes have the ability to maintain biomass for short periods of reduced clarity (Schwarz and Hawes, 1997).

Charophyte oospores failed to establish in Lake Waikare either inside or outside exclosures and sedimentation of suspended particles in the lake water was likely instrumental in the failure of this establishment. The sedimentation that had occurred (e.g. Plate 5) and the subsequent increased depth of the overlying sediment layer was considered a constraint to germination and eventual establishment of the oospores although *Chara* species can germinate in dark conditions (Takatori and Imahori, 1971, van den Berg, 1999). Reeves *et al.* (2002) in their study of Lake Waikare stated macrophyte seed banks were virtually non-existent, possibly due to burial by sporadic, heavy suspended-sediment loading (>400 g m-3). I did not detect signs of germination after 90 days exposed to submerged Lake Waikare conditions. By comparison, the tray of oospores in the incubator in the laboratory germinated by week five and their dry weight biomass increased to 0.007 g by Day 90.

Plate 5: Sedimentation on oospore pots retrieved from the exposed site (a) and the sheltered site (b) at Lake Waikare at Day 90


(a)

(b)

Chemical parameters monitored during my experiments (dissolved oxygen, pH) were found to be similar inside and outside the exclosures at both the exposed and sheltered sites in winter. This uniformity indicates the exclosures did not significantly affect the water exchange between inside and outside the exclosures. Physical parameters (Secchi depth, turbidity) similarly, were found to be uniform inside and outside the cages at each study site. Suspended sediment concentrations, however, were higher (by 25%) outside the exclosures at the exposed site and higher (by 9%) outside the exclosures at the sheltered site in winter.

Overhead conditions and wave action were likely responsible for the variability in PAR readings recorded. The euphotic depth calculated showed there was sufficient light for the plantlets to grow and in winter where the euphotic depth was higher (at the sheltered site compared to the exposed site), the plantlets grew best. In summer, the euphotic depth was generally consistent at both sites and little variation could be detected between the two sites. Chemical conditions (Dissolved Oxygen, pH) were uniform inside and outside the exclosures at both the exposed site and sheltered site in summer, Physical parameters (chlorophyll *a* PAR, Secchi depth, turbidity) were similar inside and outside the exclosures and despite variations, PAR levels were similar inside and outside the exclosures. Mean suspended sediment concentrations were uniform at the exposed site but were higher (by 9%) inside the exclosures at the sheltered site compared to outside. The highest dry weight biomass loss of charophytes (100%) was experienced in summer when turbidity was higher (by 41%) and suspended sediment concentration was higher (by 14%) compared to winter, but despite these factors the performance of protected charophyte plantlets was not adversely affected as their dry-weight biomass had increased. I conclude, therefore, that fish grazing and disturbance, and not any differences in physical or chemical conditions were the most likely cause of any dissimilarity in macrophyte biomass.

Increased macrophyte covers can lead to improved water quality when fish are removed (Shapiro and Wright, 1984; Wissel *et. al.,* 2000; McQueen, 1990) and the shift to improved water clarity from fish removal (Beklioglu *et al.,* 2003; Van Liere and Gulati, 1992) can lead to a clear-water state and if dense charophyte meadows are established (McQueen, 1991), improvements in the light climate can be dramatic (van den Bergh, 1997). A reduction of fish stocks will reduce the re-suspension of sediments (Meijer *et al.,* 1994) that this can have a positive affect on the establishment and survival of plants in lakes like Lake Waikare (He *et. al.,* 1994). Laurisden *et al.* (1994) reported that within three years of fish removal, macrophyte covers in Lake Vaeng (Denmark) had increased from 0% to 50% and two years later that cover had increased to 90%, attributed to decreased turbidity and increased light penetration (Faafeng and Mjelde, 1997). de Winton *et al.* (2002) in their assessment of the effects of fish on charophyte establishment in a shallow, eutrophic Lake Rotoroa, found charophyte dry weight biomass was greater inside exclosures compared to outside. de Winton *et al.* (2002) concluded that fish were primarily responsible for the poor performance of unprotected charophytes, either by direct disturbance or grazing (Reynolds, 1994). This appears to be the case in Lake Waikare.

3.5: Conclusions

My study showed, *C. corallina* plantlets survived and grew inside exclosures in Lake Waikare. Outside the exclosures, plantlet growth rates were restricted in winter and in summer, the plantlets were directly disturbed and totally removed and fish appear to be the mitigating factor in this disturbance and removal, impacting adversely on the potential re-establishment of submerged macrophytes in Lake Waikare. My study also showed germination of *C. corallina* oospores was severely hampered if not totally prevented by heavy sedimentation in the lake.

4: Results of Settling Experiments using Alum

4.1: Turbidity (suspended sediment concentration of 100 g l^{-1})

4.1.1: Day Zero

Mean turbidity levels of bottom waters from the Alum water column (Figure 1 a) were on average higher than mean turbidity levels from no Alum treatment after 30 minutes of settling. Mean turbidity levels peaked at ~130 NTU in the bottom waters (at time 60), then reduced steadily to ~50 NTU over time. Top waters cleared ~time 120. In the no Alum water column, mean turbidity levels were uniform over time.

4.1.2: Day 15

Mean turbidity levels of bottom waters from the Alum water column at Day 15 (Figure 1 b) were on average 50 NTU higher than mean turbidity levels from no Alum treatment. In the upper Alum water column, mean turbidity levels had reduced by ~10 NTU after starting at ~15 NTU but minimum turbidity was achieved (time 60) where levels reached their lowest values. Mean turbidity levels in the no Alum water column remained stable after beginning at ~15 NTU. The levels peaked at ~20 NTU (at time 60) then reduced ~5 NTU by time 180. Mid-column mean turbidity levels mean-while, had reduced by ~7 NTU over time.

Figure 1 (a): Mean turbidity in the Alum water column compared with mean turbidity in the no Alum water column, at Zero Interval over time (100 g l^{-1} SS conc)

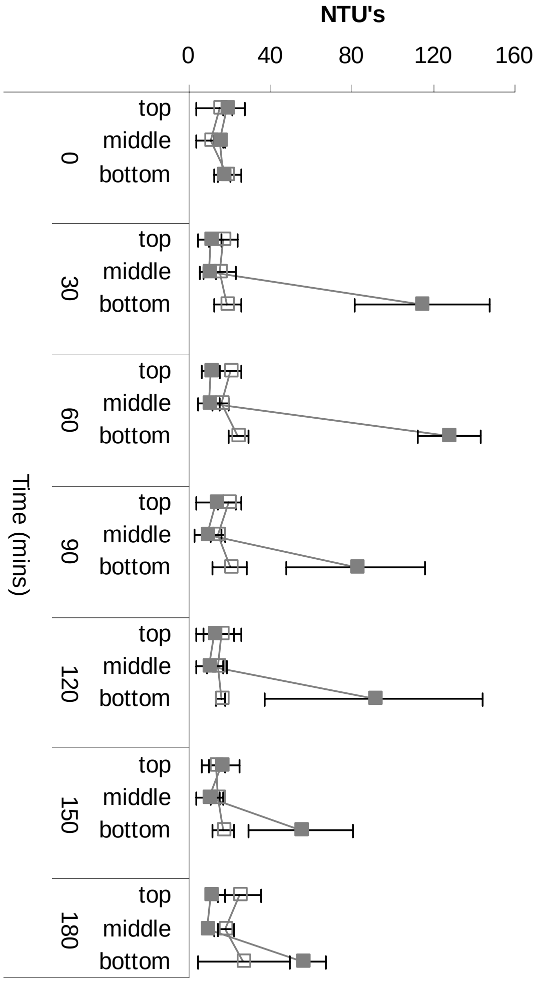
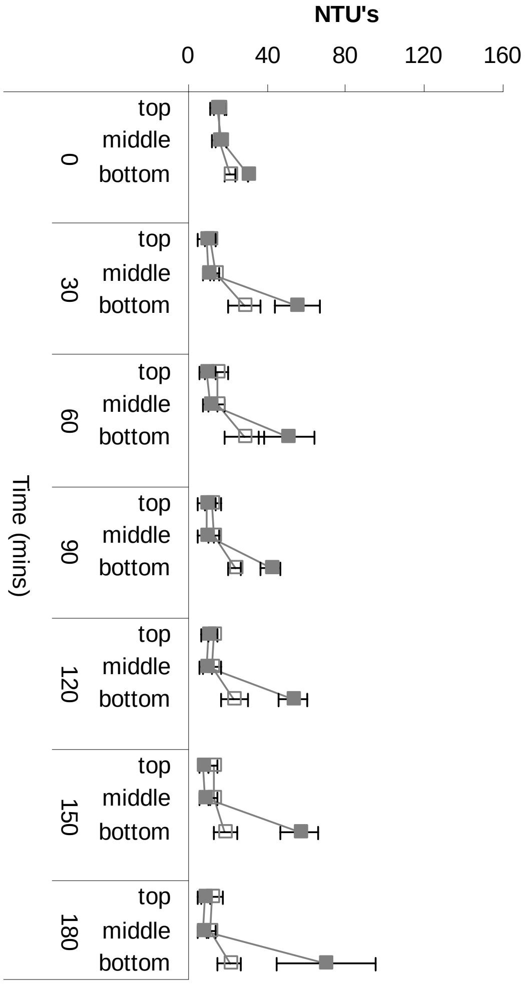


Figure 1 (b): Mean turbidity in the Alum water column compared with mean turbidity in the no Alum water column, at Day 15 over time (100 g l^{-1} SS conc)



■ Alum
□ No Alum

4.1.3: Day 30

Mean turbidity levels of bottom waters from the Alum water column increased by ~40 NTU at Day 30 (Figure 1 c) compared to no Alum treatment, peaking at ~70 NTU at time 120 in an increasing pattern. Mean turbidity levels in the upper Alum treated water column reduced by ~10 NTU to end below start levels of ~15 NTU but minimum turbidity was achieved by time 30. In the no alum water treatment, mean turbidity levels had increased by ~5 NTU over time while in the upper water column, mean turbidity levels had reduced by ~10 NTU (from ~15 NTU).

4.1.4: Day 60

Mean turbidity levels of bottom waters from the Alum water column were higher than mean turbidity levels in bottom waters from no Alum treatment. Mean turbidity levels increased from an initial ~15 NTU in an increasing fashion to peak at ~40 NTU by time 150. Mean turbidity in the upper levels of the Alum and no Alum water column, reduced uniformly from ~20 NTU to ~5 NTU by time 180 but turbidity had reached minimum levels by ~time 120 with Alum treatment.

Figure 1 (c): Mean turbidity in the Alum water column compared with mean turbidity in the no Alum water column, at Day 30 over time (100 g l^{-1} SS conc)

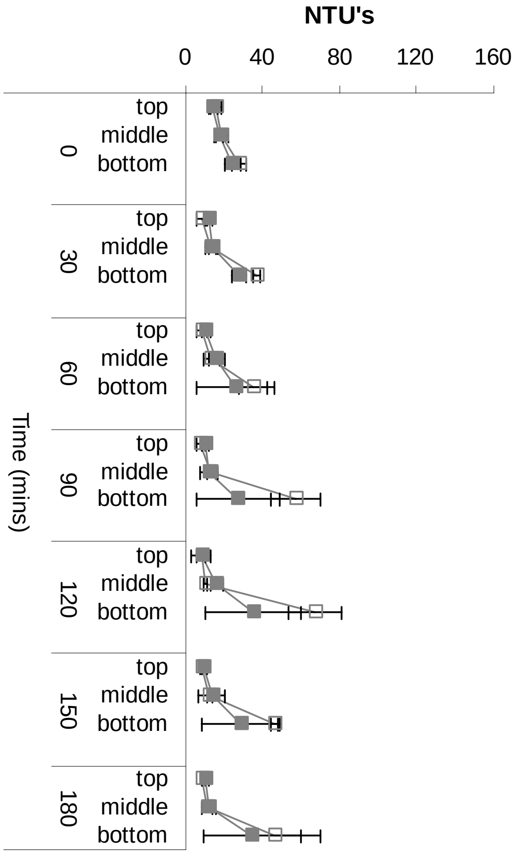
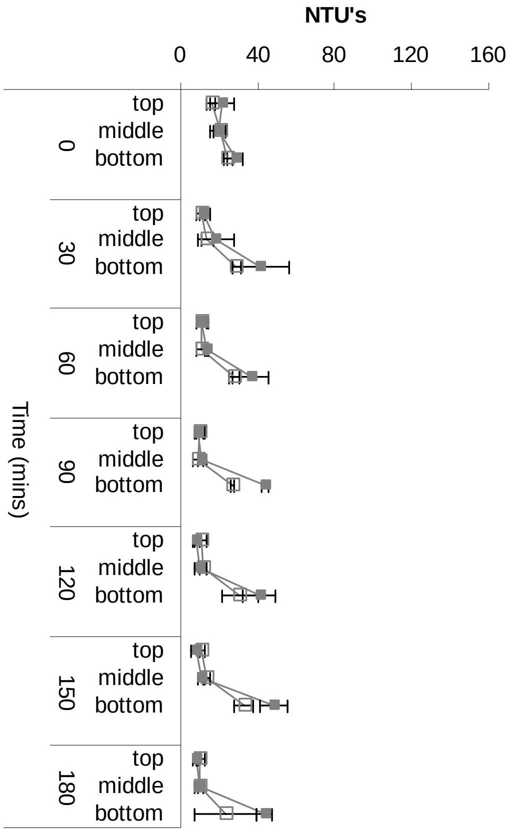


Figure 1 (d): Mean turbidity in the Alum water column compared with mean turbidity in the no Alum water column at Day 60 over time (100 g l^{-1} SS conc)



■	Alum
□	No Alum

4.2: Photosynthetically-active Radiation (suspended sediment concentration 100 g l^{-1})

4.2.1: Zero Interval

Mean PAR levels at 0 Interval (Figure 2 a) were higher in the Alum water column than no Alum water, peaking at ~30 micromoles (µm) at time 180 compared to mean PAR with no Alum treatment which peaked at 15 µm at time 150, correlating to where turbidity had reached minimum levels (Figure 1 a).

4.2.2: Day 15

Mean PAR levels at Day 15 (Figure 2 b) had increased to 60 µm (by time 120) in the Alum water column then increased markedly to ~100 µm (by time 180). By comparison, mean PAR levels in no Alum treatment remained stable at an average 30 µm over time. Increases in PAR at time 90, correlates to minimum turbidity being achieved (Figure 1 b).

Figure 2 (a): Mean PAR in the Alum water column compared with mean PAR in the no Alum water column at Zero Interval over time (100 g l^{-1} SS conc)

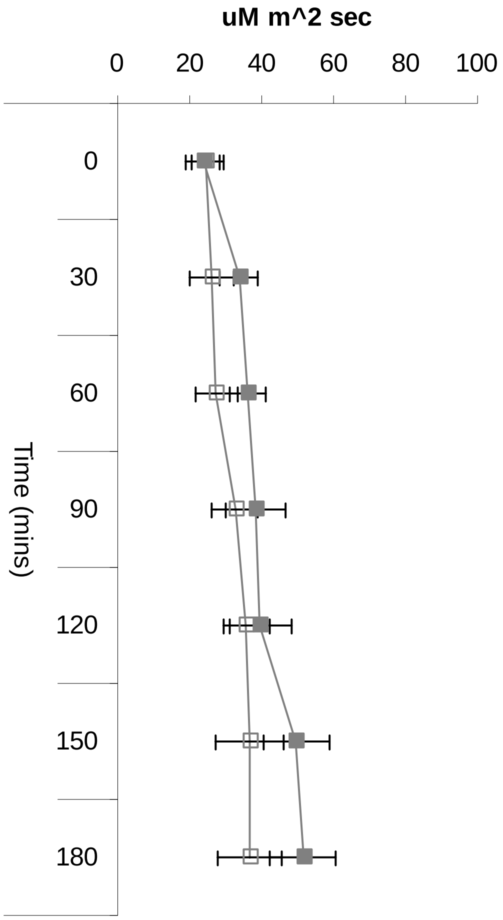
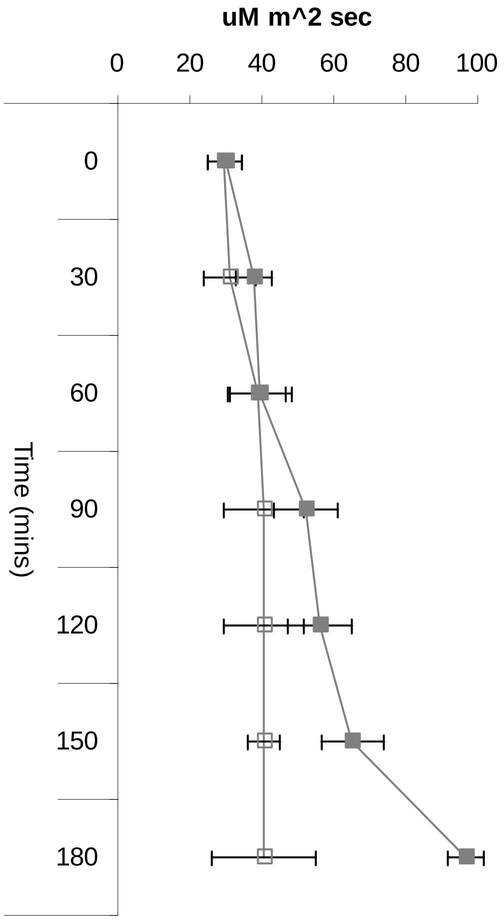


Figure 2 (b): Mean PAR in the Alum water column compared with mean PAR in the no Alum water column at Day 15 over time (100 g l^{-1} SS conc)



■ Alum
□ No Alum

4.2.3: Day 30

Mean PAR levels increased to ~65 µm (by time 90) in the Alum water column at Day 30 (Figure 2 c) then decreased to ~60 µm by time 180. Mean PAR levels in no Alum treatment increased in an increasing pattern to peak at ~50 µm (at time 150) then reduced to ~45 µm by time 180.

4.2.4: Day 60

Mean PAR levels at Day 60 (Figure 2 d) increased to ~40 µm in the Alum water column (by time 90), correlating to minimum levels in the turbidity result (Figure 1 d) at the same time.. Mean PAR levels in the no Alum treatment increased from ~20 µm (time 0) to ~30 µm at time 180.

Figure 2 (c): Mean PAR in the Alum water column compared with mean PAR in the no Alum water column at Day 30 over time (100 g l^{-1} SS conc)

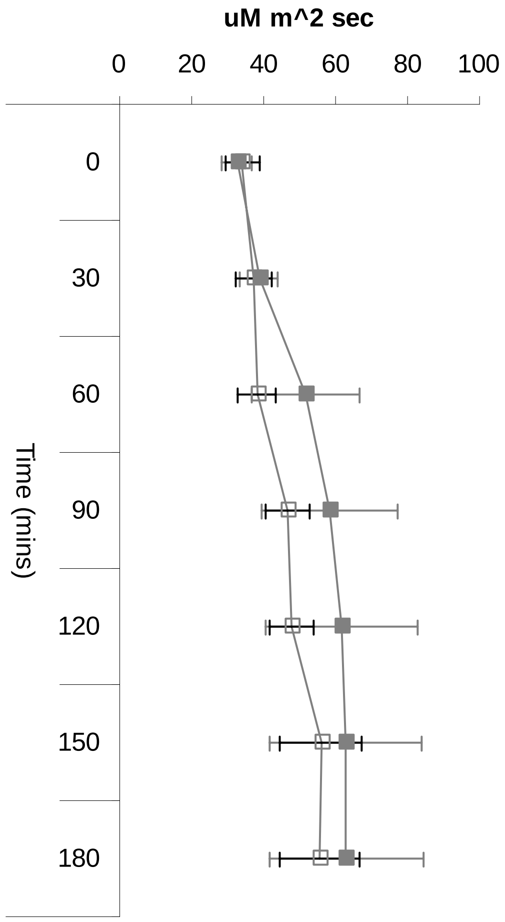
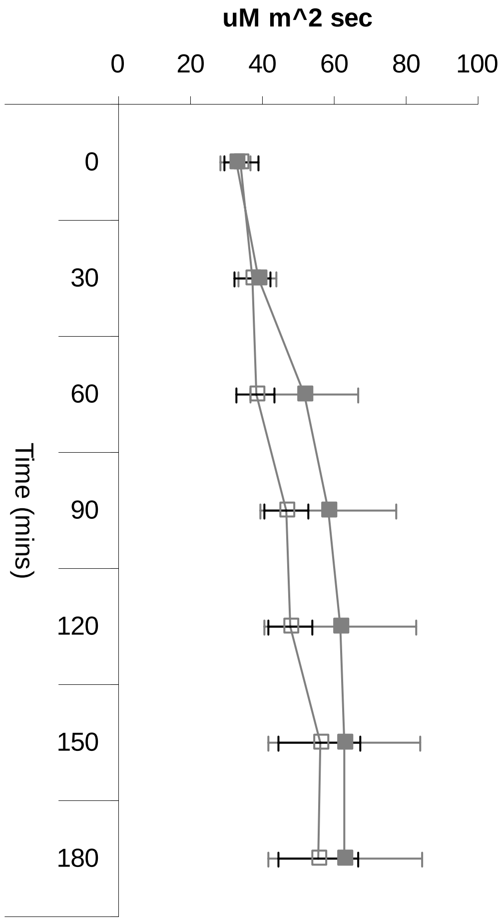


Figure 2 (d): Mean PAR in the Alum water column compared with mean PAR in the no Alum water column at Day 60 over time (100 g l^{-1} SS conc)

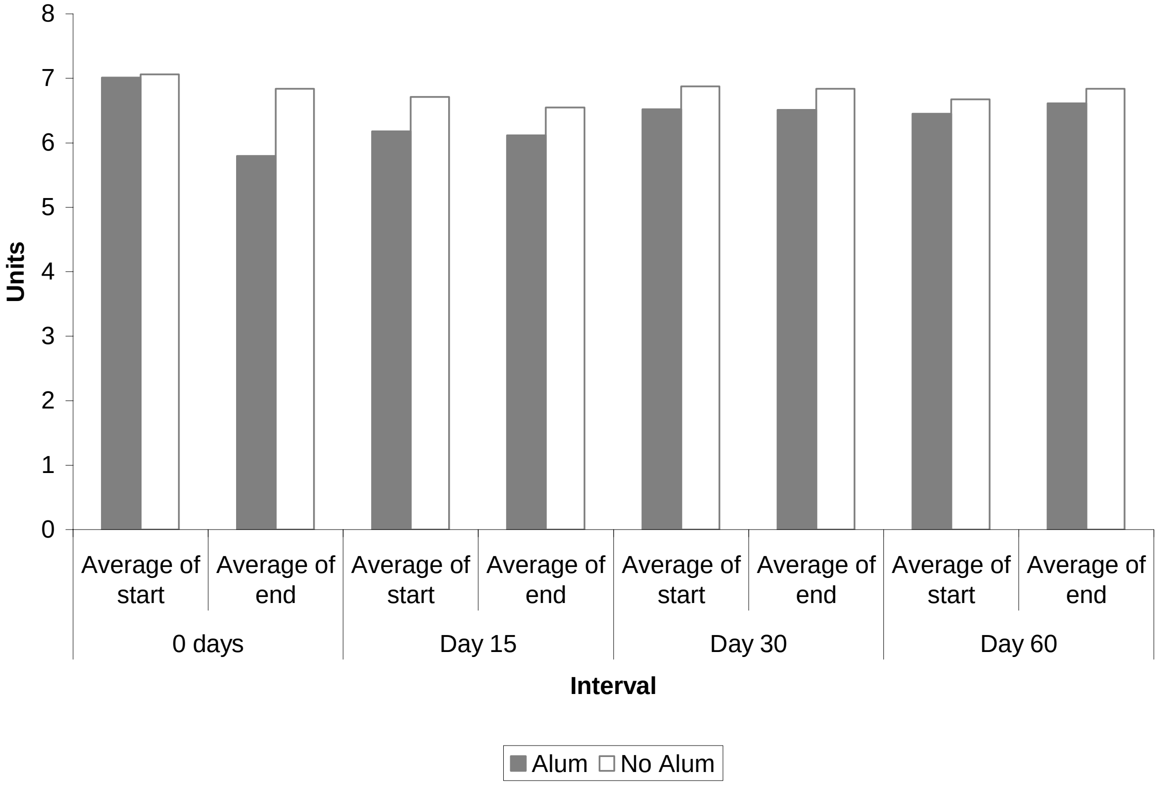


■	Alum
□	No Alum

4.3: pH (suspended sediment concentration 100 g l^{-1})

Mean pH levels in Alum treatments (Figure 3) were lower (by ~1.3 units) at the end of the 0 Interval experiment compared to the beginning. However, mean pH for the Day 15 and 30 experiments were similar with mean pH at Day 60 marginally higher at the end of the experiment than at the start. By comparison, mean pH for the no Alum treatments were uniform between the start and end of the experiments for the four intervals.

Figure 3: Mean pH in the Alum water treatment compared with mean pH of the no Alum treatment, pre- and post-experiment over four, 15 day intervals (100 g l^{-1} SS conc)



4.4: Suspended Sediment Concentration (suspended sediment concentration 100 g l^{-1})

4.4.1 Zero Interval

Mean SS concentrations of bottom waters from the Alum water column on Day 0 (Figure 4 a) were higher on average than mean bottom waters from the no Alum water column, particularly to time 90. Differences in settling from time 120 had ceased for both the Alum and no Alum treatments from time 120. In the upper water column, mean SS concentration on average, was higher in the Alum water column than the no Alum water column, particularly to time 90

4.4.2: Day 15

Mean SS concentrations of bottom waters were consistently higher on average from the Alum water column than the no Alum water column at Day 15 (Figure 4 b) over time. In the upper water column, mean SS concentration of Alum treatment was generally higher than no Alum treatment with the lowest reading (at time 60) indicating minimum turbidity occurred there.

Figure 4 (a): Mean suspended sediment concentration in the Alum water column compared with mean suspended sediment concentration in the no Alum water column at Zero Interval over time (100 g l^{-1} SS conc)

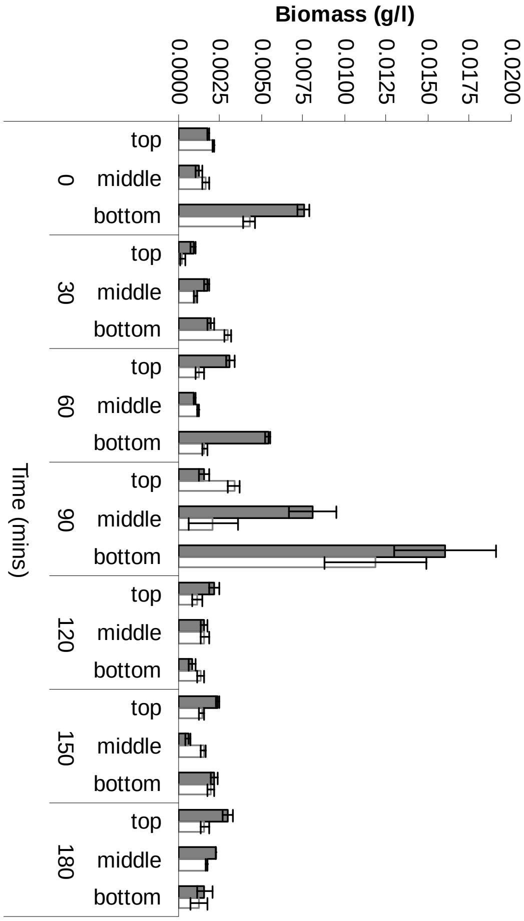
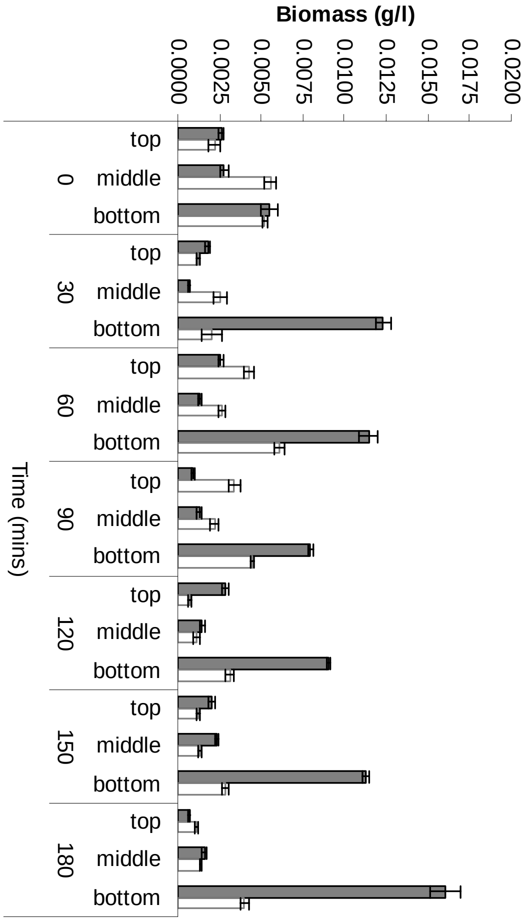


Figure 4 (b): Mean suspended sediment concentration in the Alum water column compared with mean suspended sediment concentration in the no Alum water column at Day 15 over time (100 g l^{-1} SS conc)



4.4.3: Day 30

Mean SS concentration of bottom waters were consistently higher from the Alum water column than from the no Alum water column at Day 30 (Figure 4 c) with peak settling occurring between time 90 and time 120. In the upper water column, mean SS concentration was generally higher for the Alum treatment compared to the no treatment with a slight pattern of decline in settling emerging with values equal or lower for Alum treatment than no Alum treatment, over time. Minimum turbidity occurred at the lowest level in Alum treatment (~time 30).

4.4.4: Day 60

Mean SS concentration of bottom waters from the Alum water column were consistently higher on average than those from no Alum water at Day 60 (Figure 4 d) especially from time 90. Mean SS concentration in the upper water column was higher on average in Alum treatment to time 90 where minimum turbidity was recorded.

Figure 4 (c): Mean suspended sediment concentration in the Alum water column compared with mean suspended sediment concentration in the no Alum water column at Day 30 over time (100 g l^{-1} SS conc)

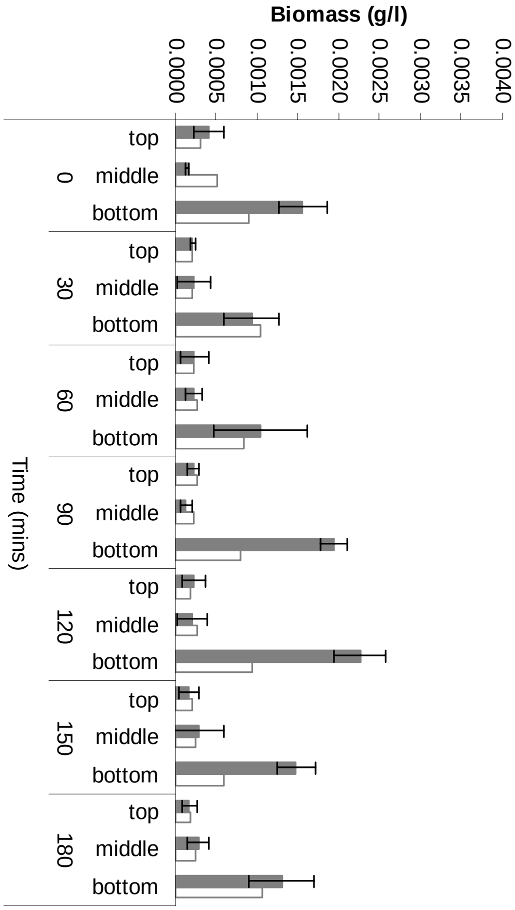
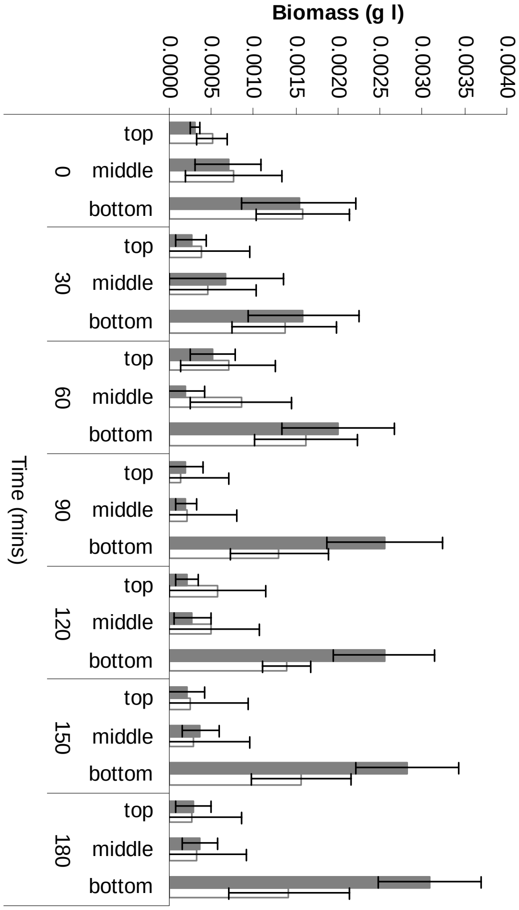


Figure 4 (d): Mean suspended sediment concentration in the Alum water column compared with mean suspended sediment concentration in the no Alum water column at Day 60 over time (100 g l^{-1} SS conc)



■	Alum
☐	No Alum

4.5: Flocculation (suspended sediment concentration 200 g l^{-1})

4.5.1: Turbidity

4.5.1.1: Zero Interval

Mean turbidity levels of bottom waters from the Alum water column was consistently higher (~140 NTU at time 30) than bottom samples from the no Alum water column at 0 Interval (Figure 5 a). From time 30, there is pattern of reducing mean turbidity levels of bottom waters treated with Alum to time 180. Mean turbidity in the upper Alum water column indicate a pattern of decreasing values (from ~30 NTU to ~5 NTU) over time, reaching its lowest levels (at ~time 120) where minimum turbidity in top waters occurred.

4.5.1.2 Day 15

Mean turbidity levels of bottom waters from the Alum water column was higher by ~35 NTU (at time 180) than mean turbidity levels of bottom waters without Alum treatment at Day 15 (Figure 5 b). Mean turbidity in the upper levels of the water column with Alum treatment reduced by ~20 NTU to time 150 where minimum turbidity occurred, compared to a reduction of ~10 NTU with no Alum treatment

Figure 5 (a): Mean turbidity in the Alum water column compared with mean turbidity in the no Alum water column at Zero Interval over time (100 g l^{-1} SS conc)

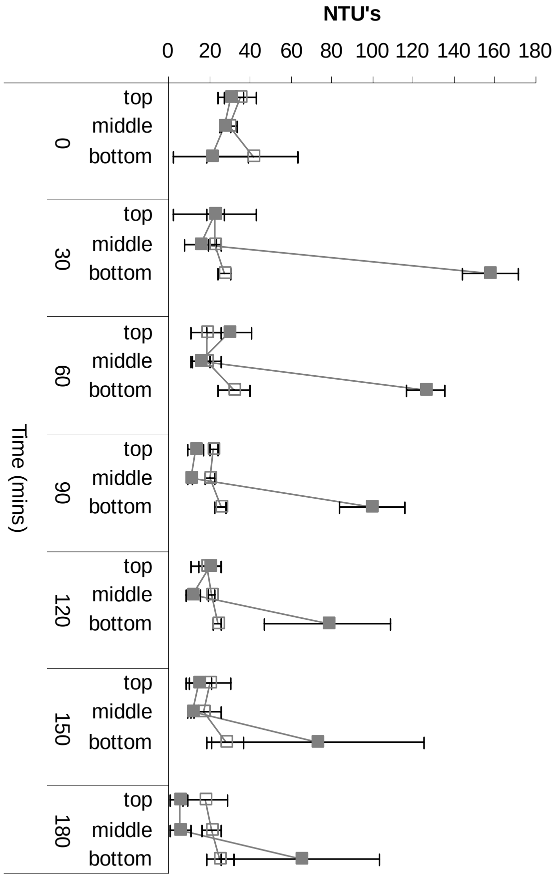
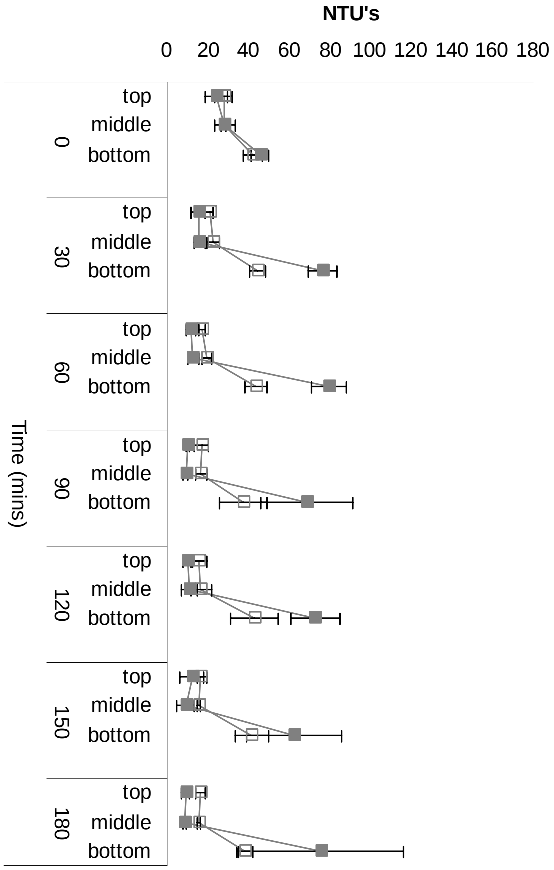


Figure 5 (b): Mean turbidity in the Alum water column compared with mean turbidity in the no Alum water column at Day 15 over time (100 g l^{-1} SS conc)



■	Alum
☐	No Alum

4.5.1.3: Day 30

Mean turbidity levels of bottom waters from the Alum water column was marginally higher than mean turbidity levels in the no Alum water column at Day 30 (Figure 5 c). Mean turbidity levels with Alum treatment occurred at time 30 where the greatest difference (~40 NTU) occurred From this time, mean turbidity levels reduced from 70 NTU to 40 NTU. Mean-while mean turbidity levels in the upper water column had decreased from 20 NTU to ~5 NTU for both Alum and no Alum treatments with minimum turbidity occurring at time 30.

4.5.1.4: Day 60

Mean turbidity levels at Day 60 (Figure 5 d) mirrored mean turbidity levels for Day 30 except maximum mean turbidity occurred at time 60 (cf time 30 at Day 30). Similarly, this was where the greatest difference (~20 NTU) occurred, and from where turbidity reduced (from ~65 NTU) to ~60 NTU to time 180. Mean turbidity levels in the upper water column indicates minimum turbidity occurred at ~time 90. Both treatments decreased from an initial mean level of ~25 NTU to ~10 NTU at time 180.

Figure 5 (c): Mean turbidity in the Alum water column compared with mean turbidity in the no Alum water column at Day 30 over time (100 g l^{-1} SS conc)

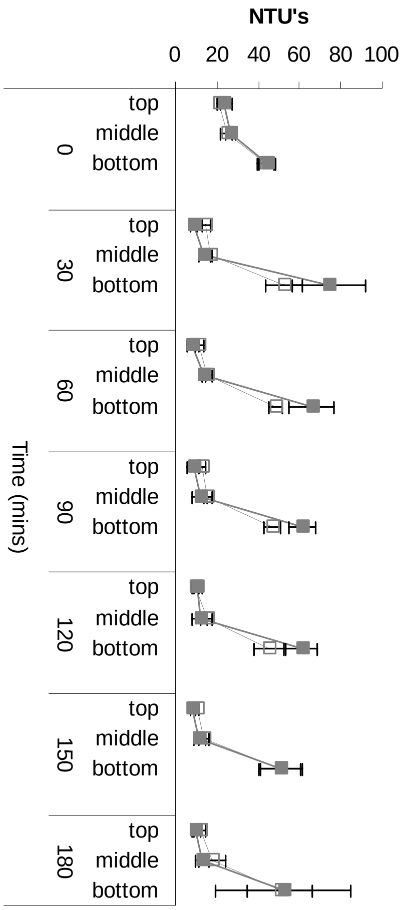
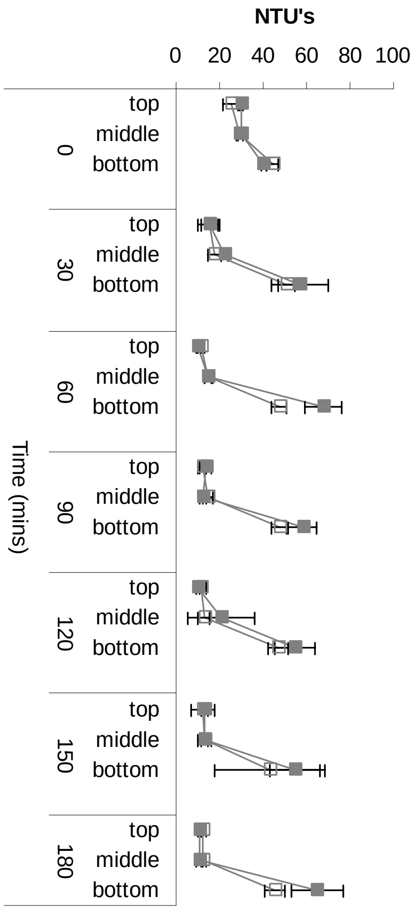


Figure 5 (d): Mean turbidity in the Alum water column compared with mean turbidity in the no Alum water column at Day 60 over time (100 g l^{-1} SS conc)



▨	Alum
☐	No Alum

4.6: Photosynthetically-active Radiation (suspended sediment concentration 200 g/l^{-1})

4.6.1 Zero Interval

Mean PAR levels was consistently higher in the Alum water column than the no Alum water column at 0 Interval (Figure 6 a). At time 120, settling rates for both treatments appeared to converge after which mean PAR levels in no Alum treatment increased (~7 µm to ~12 µm) but while no Alum treatment stabilised, Alum treatment increased sharply to ~28 µm (time 150) where a correlation with minimum turbidity (Figure 5 a) exists.

4.6.2: Day 15

Mean PAR levels in the Alum water column at Day 15 (Figure 6 b) increased in increments of ~2 µm (from ~12 µm) to time 150 when it increased sharply to 28 µm (at time 180). By comparison, mean PAR levels in no Alum treatment remained stable (~15 µm) to time 120 where it increased to ~20 µm.

Figure 6 (a): Mean PAR in the Alum water column compared with mean PAR in the no Alum water column at Zero Interval over time (200 g l^{-1} SS conc)

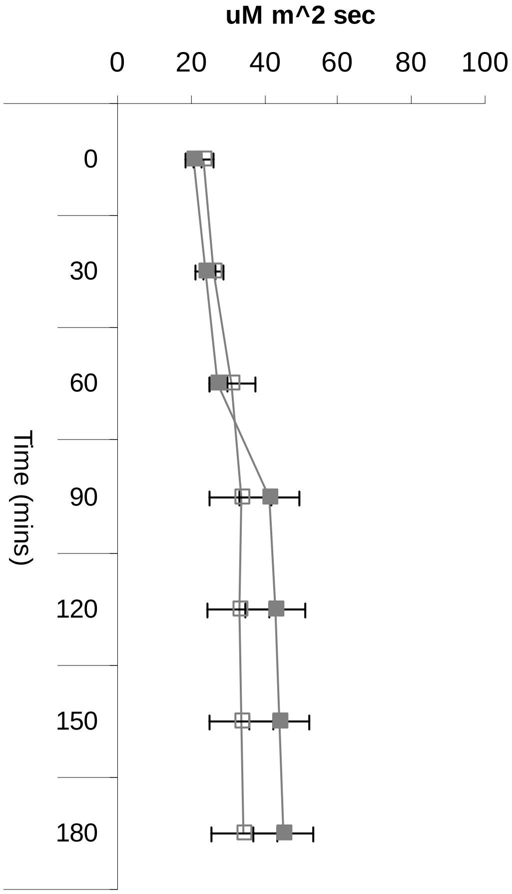
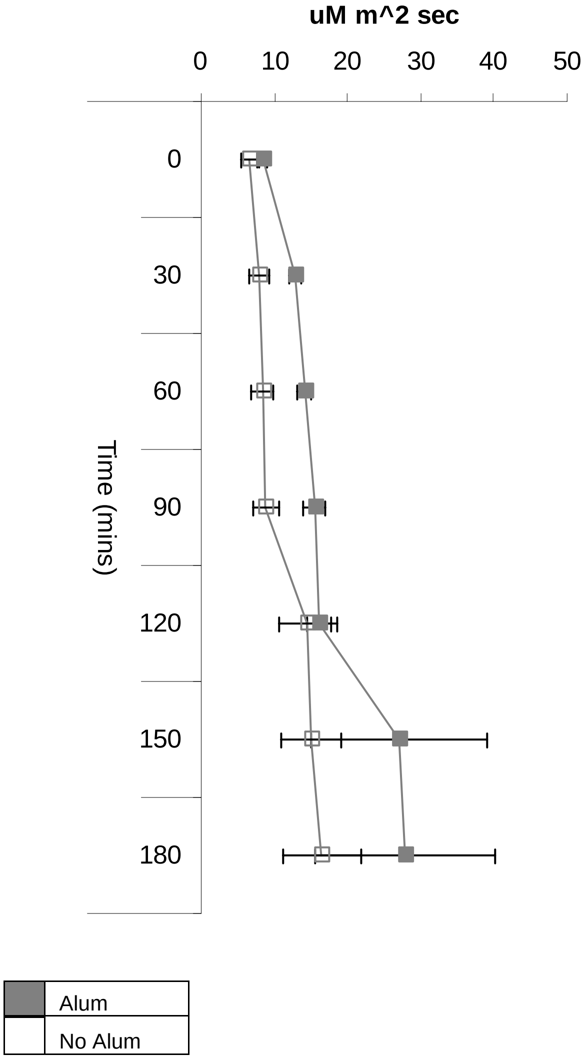


Figure 6 (b): Mean PAR in the Alum water column compared with mean PAR in the no Alum water column at Day 15 over time (200 g l^{-1} SS conc)



4.6.3: Day 30

Mean PAR levels in the Alum water at Day 30 (Figure 6 c) increased (by ~7 µm) from starting levels of ~15 µm to time 60, correlating closely with minimum turbidity (Figure 5 c) and stabilising to time 180. By comparison, mean PAR levels in no Alum remained stable at ~15 µm over time.

4.6.4: Day 60

Mean PAR levels in the Alum water column at Day 60 (Figure 6 d) mirrored the no Alum water column to time 60 Mean PAR levels traced similar paths (to time 60) then diverged with mean PAR levels for Alum treatment increasing sharply (by 14 µm) to peak at ~32 µm at time 90 (minimum turbidity, Figure 5 d) where it remained to time 180. Concurrently, mean PAR levels in no Alum treatment increased by ~5 µm (to ~20 µm) at time 90 where it remained to time 180.

Figure 6 (c): Mean PAR in the Alum water column compared with mean PAR in the no Alum water column at Day 30 over time (200 g l^{-1} SS conc)

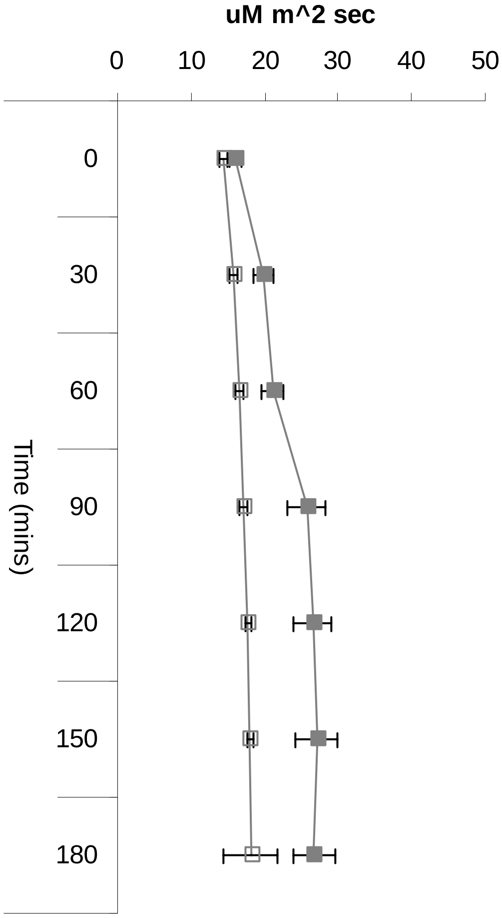
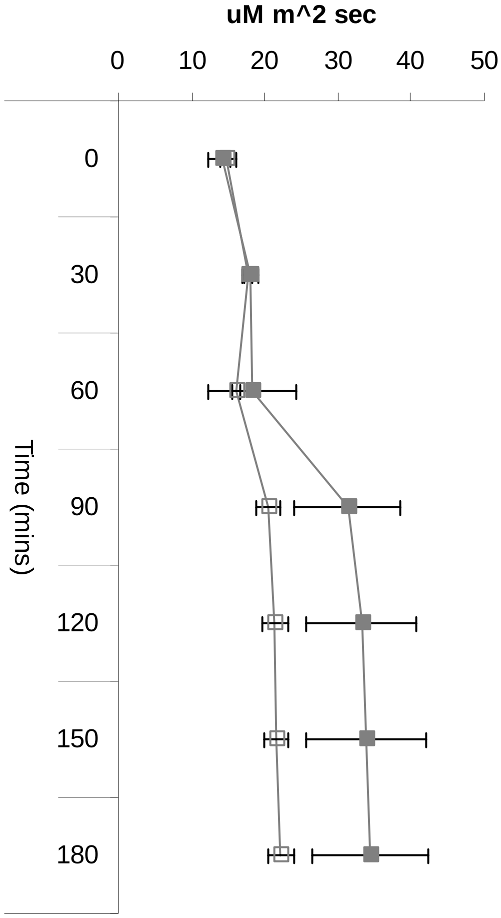


Figure 6 (d): Mean PAR in the Alum water column compared with mean PAR in the no Alum water column at Day 60 over time (200 g l^{-1} SS conc)

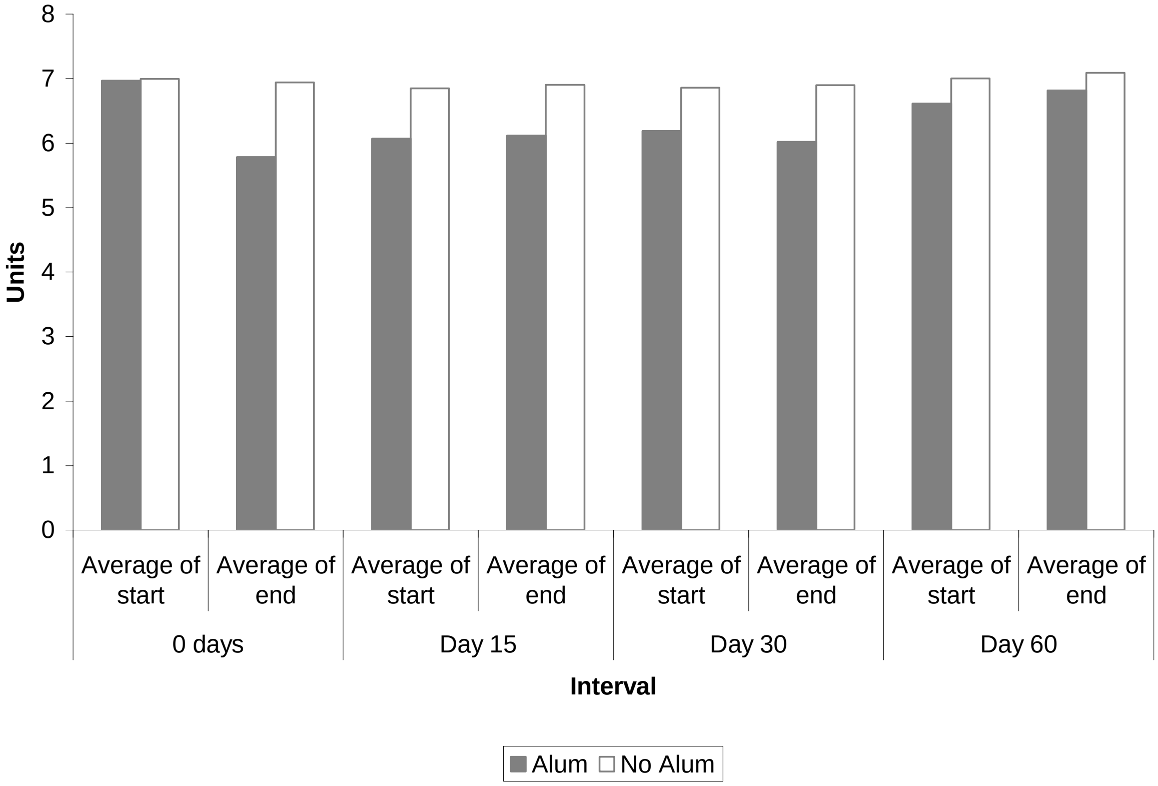


■	Alum
□	No Alum

4.7: pH (suspended sediment concentration 200 g l^{-1})

Mean pH levels at the end of the 0 Interval experiment (Figure. 7) were lower (by ~1.25 units) than at the start while mean pH levels at Day 15 were similar between the start and end. Mean pH levels at Day 30 had reduced from start levels to end ~0.25 units lower than mean start levels, however, mean pH levels at Day 60, had increased by a similar margin (~0.25 units). By comparison, mean pH levels in the no Alum water column were uniform between the start and end of the experiments across all intervals.

Figure 7: Mean pH in the Alum water column compared with mean pH in the no Alum water column pre- and post-experiment over four 15 day intervals (200 g l^{-1} conc)



4.8: Suspended Sediment Concentration (suspended sediment concentration 200 g l^{-1})

4.8.1: Zero Interval

Mean SS concentrations in the Alum water column was consistently higher than in no Alum water at 0 Interval (Figure 8 a). Mean SS concentration of bottom waters with Alum treatment peaked at time 90, while mean SS concentrations with no Alum treatment peaked later at time 120. In the upper levels of the water column, a reducing pattern of mean SS concentrations with Alum treatment is evident. Mean concentration reduced from an initial 0.0075 g l^{-1} (at time 0) to near 0 at time 180 however, mean SS concentrations in water with no Alum treatment showed reduced response rates across time compared to Alum treatment.

4.8.2: Day 15

Mean SS concentration of bottom waters from Alum treatment at Day 15 (Figure 8 b) peaked (at 0.0325 g l^{-1}) at time 30 then declined (to 0.011 g l^{-1}) by time 180. By comparison, mean SS concentration of bottom waters with no Alum treatment remained stable (at ~0.015 g l^{-1}). In the upper water column, mean SS concentration in Alum treated water was consistently lower than mean SS concentrations in no Alum water, with minimum turbidity occurring in Alum treatment at ~time 150.

Figure 8 (a): Mean suspended sediment concentration in the Alum water column compared with mean suspended sediment concentration in the no Alum water column at Zero Interval over time (200 g l^{-1} SS conc)

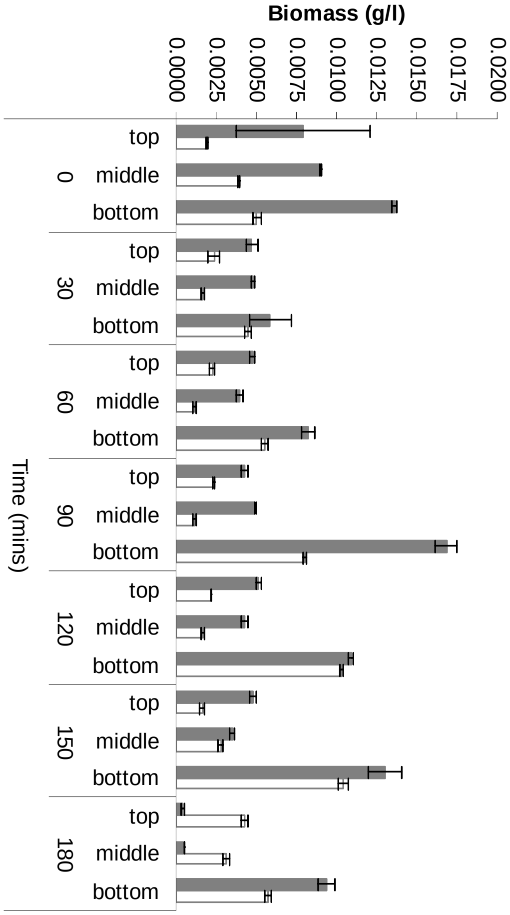
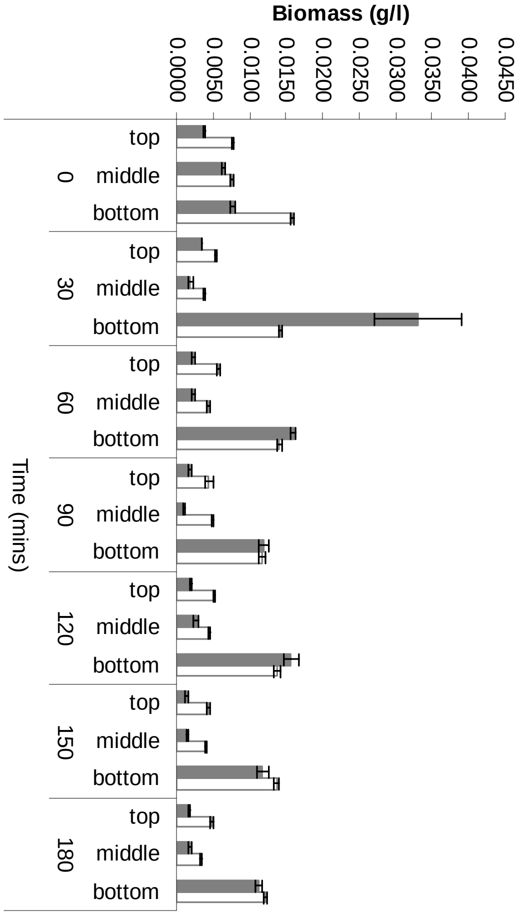


Figure 8 (b): Mean suspended sediment concentration in the Alum water column compared with mean suspended sediment concentration in the no Alum water column at Day 15 over time (200 g l^{-1} SS conc)



▨	Alum
☐	No Alum

4.8.3: Day 30

Mean SS concentrations of bottom samples from both Alum and no Alum treatments at Day 30 (Figure 8 c), peaked at mean SS concentrations of 0.0025 g l^{-1} and 0.003 g l^{-1} respectively (at time 30). Mean concentrations stabilised at these levels until time 60 when they reduced to 0.0009 g l^{-1} to 0.0015 g l^{-1} respectively by time 180. Mean SS concentrations in the upper levels show a pattern of reducing values from initial highs (0.001 g l^{-1}) in Alum treated water at 0 time, to half that value by time 180 (minimum turbidity). Mean SS concentrations in Alum treatment were generally equal to, or less than mean SS concentrations of water with no Alum treatment.

4.8.4: Day 60

Mean SS concentration of bottom waters from both the Alum and no Alum treatments at Day 60 (Figure 8 d), peaked at 0.004 g l^{-1} and 0.003 g l^{-1} respectively (time 30 and time 60). From these points, mean SS concentrations reduced for both Alum and no Alum treatments to 0.0035 g l^{-1} and 0.0025 g l^{-1} respectively. In the upper levels of the water column, a reducing pattern of mean SS concentrations in both the Alum and no Alum water is evident, with Alum levels indicating minimum turbidity occurring at time 60.

Figure 8 (c): Mean suspended sediment concentration in the Alum water column compared with mean suspended sediment concentration down the no Alum water column at Day 30 over time (200 g l^{-1} SS conc)

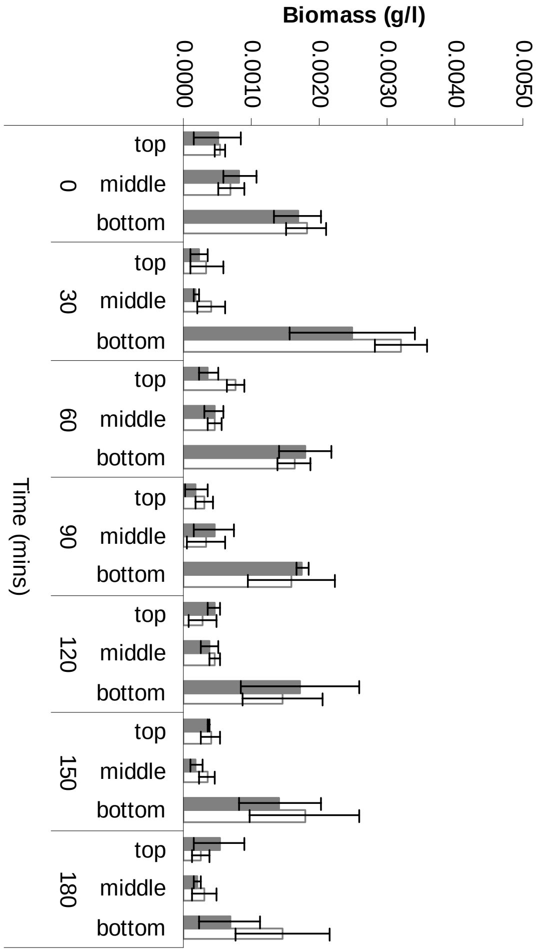
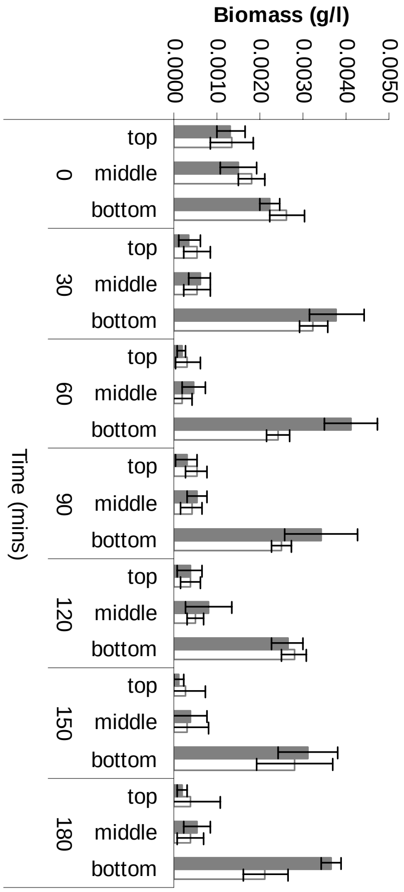


Figure 8 (d): Mean suspended sediment concentration in the Alum water column compared with mean suspended sediment concentration down the no Alum water column at Day 60 over time (200 g l^{-1} SS conc)



▇	Alum
□	No Alum

4.9: Flocculation (suspended sediment concentration 300 g l^{-1})

4.9.1: Turbidity

4.9.1.1: Zero Interval

Mean turbidity levels in Alum treated water were higher at the top of the column (by 20 NTU) than at the top of the no Alum water column at 0 Interval (Figure 9 a). Minimal settling occurred with mean turbidity levels remaining subdued (compared to previous experiments) to time 180. Mean turbidity of bottom waters with no Alum were lower (by 20 NTU) than mean turbidity of bottom waters treated with Alum. Mean turbidity in no Alum water had increased (to ~40 NTU) by time 120 while mean turbidity of bottom waters had reduced from 10 NTU (at time 0) to ~5 NTU by time 180

4.9.1.2: Day 15

Mean turbidity at Day 15 (Figure 9 b) shows a similar pattern to 0 Interval (Figure 9 a). Mean turbidity levels were higher at the top of the Alum water column (by ~30 NTU's) than at the top of the no Alum water column. Mean turbidity of bottom waters peaked at 60 NTU, a level that was maintained to time 120 where it reduced (by 20 NTU) to 40 NTU by time 180. In the upper levels of the water column meanwhile, mean turbidity had reduced from 30 NTU (at time 0) to minimum levels (15 NTU) by time 60 (minimum turbidity) stabilising there to time 180. By comparison, mean turbidity of bottom waters from the no Alum treatment peaked at~25 NTU (time 60) then reduced to ~10 NTU by time 180. Mean turbidity in the upper levels of the no Alum water column remained stable and uniform for the duration of the experiment.

Figure 9 (a): Mean turbidity in the Alum water column compared with mean turbidity in the no Alum water column at Zero Interval over time (300 g l^{-1} SS conc)

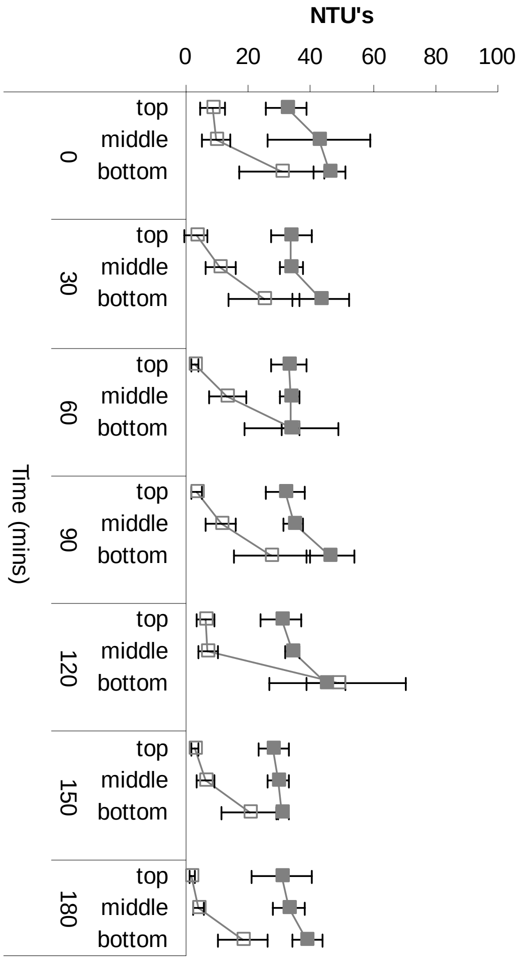
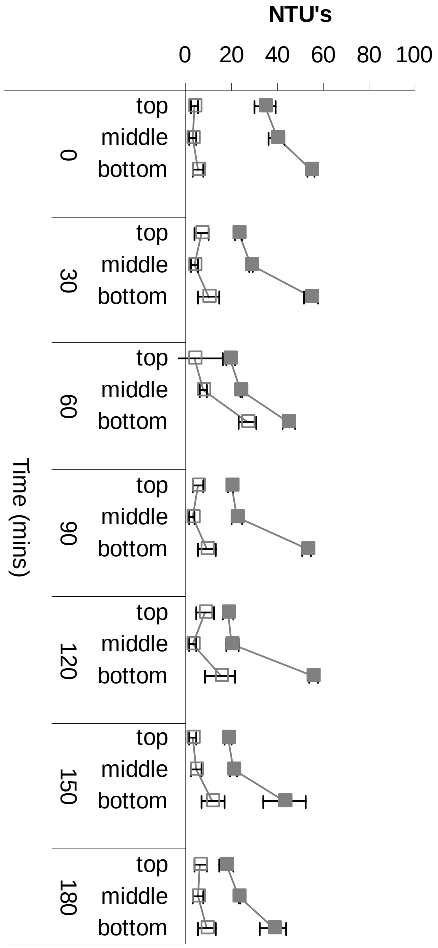


Figure 9 (b): Mean turbidity in the Alum water column compared with mean turbidity in the no Alum water column at Day 15 over time (300 g l^{-1} SS conc)



■	Alum
□	No Alum

4.9.1.3: Day 30

Mean turbidity of bottom samples from the Alum water column remained at peak levels (~60 NTU) between time 0 and time 120, reducing to ~50 NTU at time 180. Mean turbidity in the upper levels of the water column reduced from ~40 NTU at (time 0) reaching minimum levels (~5 NTU, minimum turbidity) by ~time 60. Mean turbidity levels in no Alum treatment peaked at ~15 NTU (time 30), thereafter reducing to ~10 NTU by time 180.

4.9.1.4: Day 60

Mean turbidity levels of bottom waters from the Alum water column peaked at ~80 NTU (time 30) then reduced to ~60 NTU by time 180. Mean turbidity in the upper levels of the Alum water column, reduced from ~35 NTU (time 0) to ~10 NTU by time 180. Mean turbidity levels in bottom waters from the no Alum water column peaked at ~15 NTU (time 30) and by time 180, mean turbidity had reduced to ~8 NTU. Mean turbidity in the upper column reached minimum levels (minimum turbidity) by time 60 and remained stable there (at ~5 NTU) throughout the experiment.

Figure 9 (c): Mean turbidity in the Alum water column compared with mean turbidity in the no Alum water column at Day 30 over time (300 g l^{-1} SS conc)

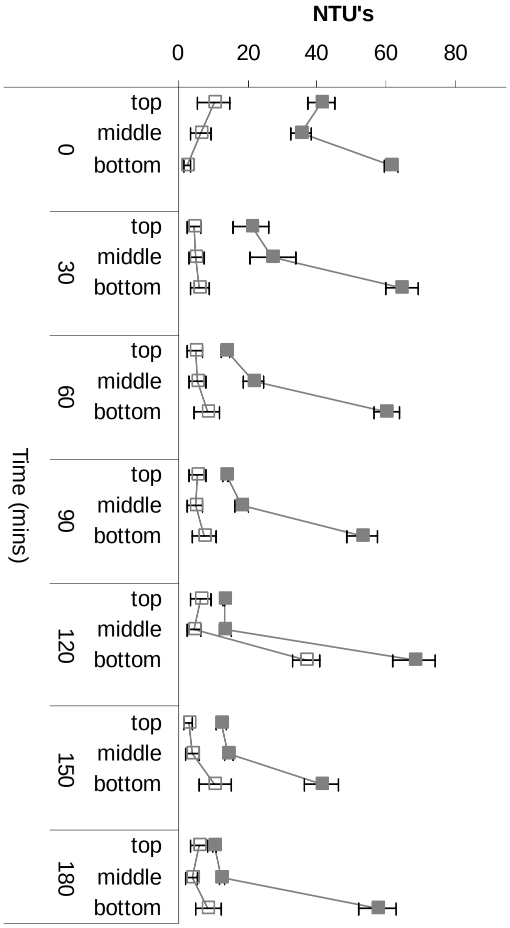
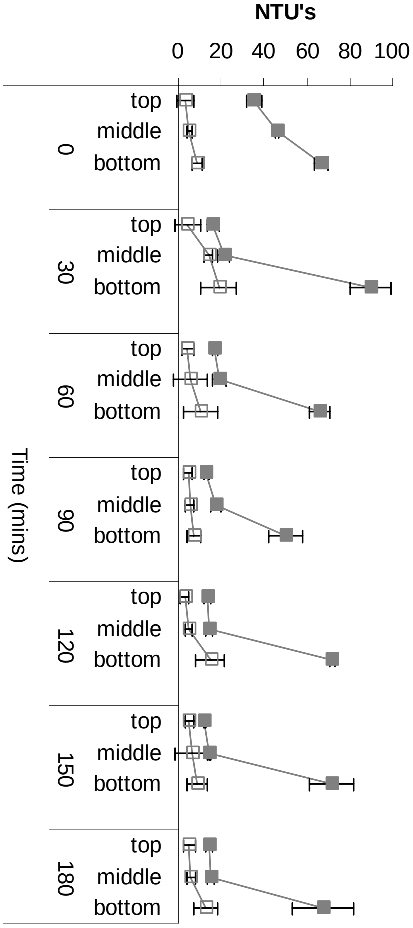


Figure 9 (d): Mean turbidity in the Alum water column compared with mean turbidity in the no Alum water column at Day 60 over time (300 g l^{-1} SS conc)



■ Alum
□ No Alum

4.10: Photosynthetically-active Radiation (suspended sediment concentration 300 g l^{-1})

4.10.1: Zero Interval

Mean PAR levels in the Alum water column increased by 2 µm peaking at 10 µm (at time 120) at 0 Interval (Figure 10 a) stabilising at this level to time 180. Mean PAR levels in no Alum water was stable (at ~2 µm) until time 120 when mean PAR levels increased sharply to ~7 µm (time 150) where it remained to time 180.

4.10.2: Day 15

Mean PAR levels in the Alum water column at Day 15 (Figure 10 b) increased gradually (by ~3 µm) to time 180 while mean PAR in no Alum water column increased similarly (by ~2 µm) to time 180. The increase at time 120 correlates to minimum turbidity occurring at ~time 120 (Figure 9 b).

Figure 10 (a): Mean PAR in the Alum water column compared with mean PAR in the no Alum water column at Zero Interval over time (300 g l^{-1} SS conc)

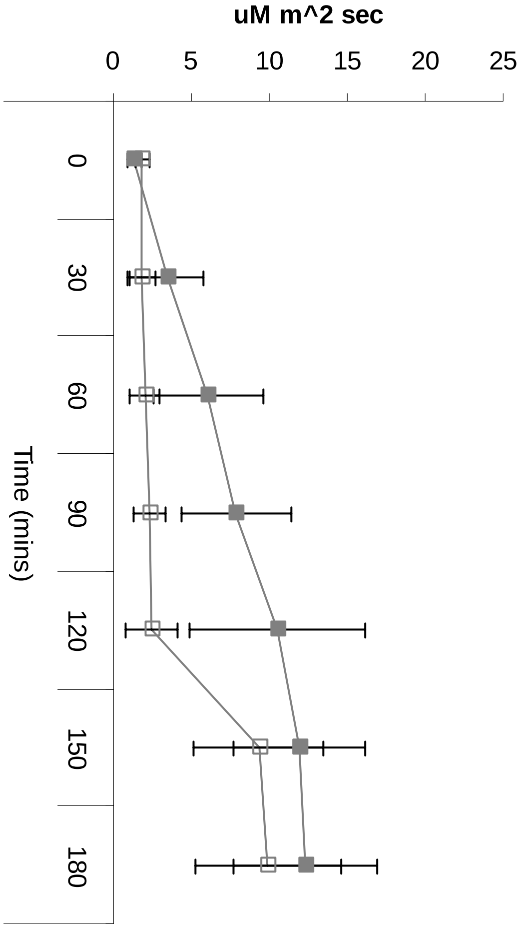
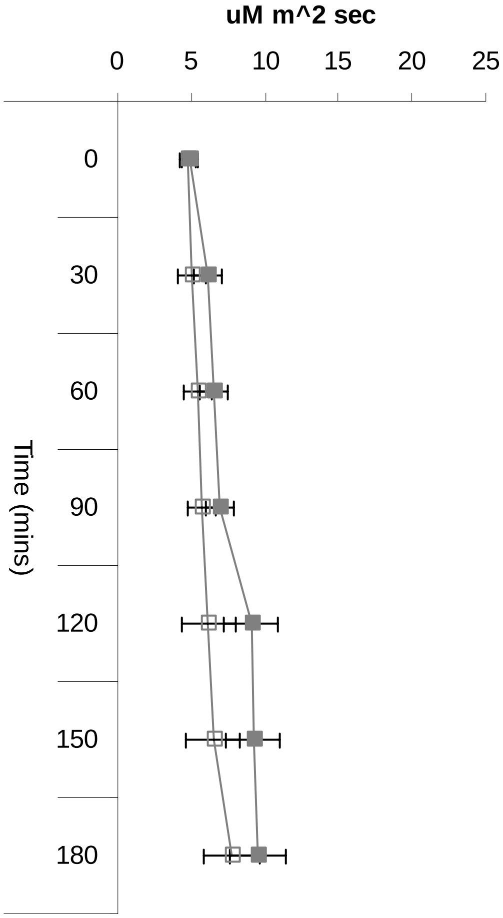


Figure 10 (b): Mean PAR in the Alum water column compared with mean PAR in the no Alum water column at Day 15 over time (300 g l^{-1} SS conc)



■	Alum
☐	No Alum

4.10.3: Day 30

Mean PAR levels in Alum and no Alum waters (Figure 10 c) were similar at ~7 µm (to time 60). Mean PAR levels in Alum water increased sharply from time 60 (correlating to minimum turbidity, Figure 9 c) to peak at 24 µm at time 150, then reduced to ~15 µm at time 180. Mean PAR in the no Alum water column overall, increased by ~2 µm from an initial mean level of 6 µm (time 0).

4.10.4: Day 60

Mean PAR levels at Day 60 (Figure 10 d) in both the Alum and no Alum water columns, increased in similar fashion from initial levels of ~6 µm (time 0) increasing to ~10 NTU by time 180.

Figure 10 (c): Mean PAR in the Alum water column compared with mean PAR in the no Alum water column at Day 30 over time (300 g l^{-1} SS conc)

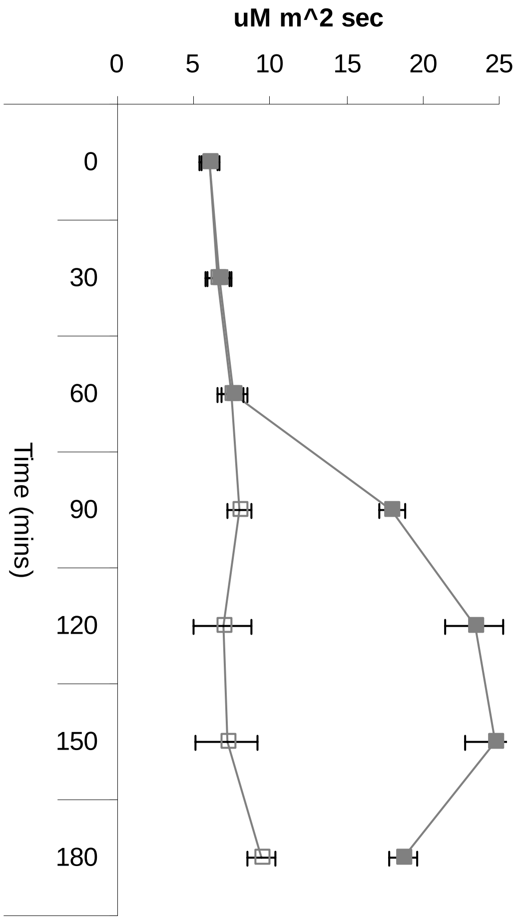
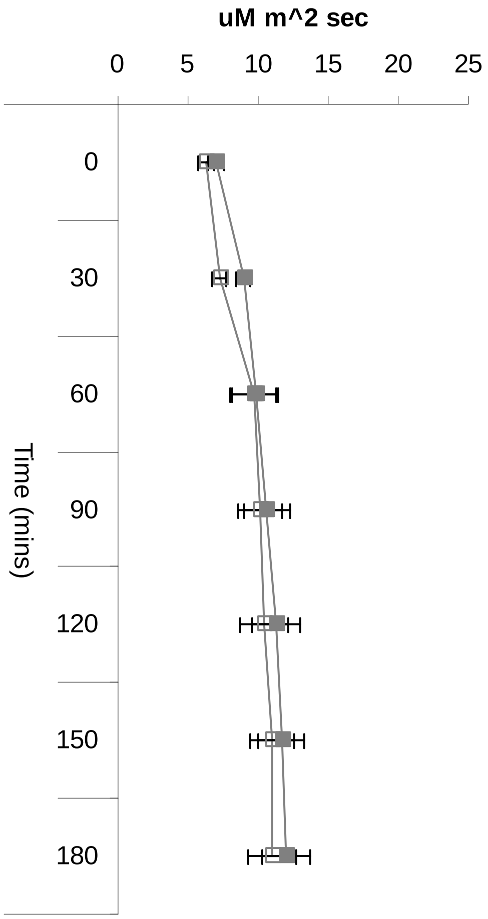


Figure 10 (d): Mean PAR in the Alum water column compared with mean PAR in the no Alum water column at Day 60 over time (300 g l^{-1} SS conc)

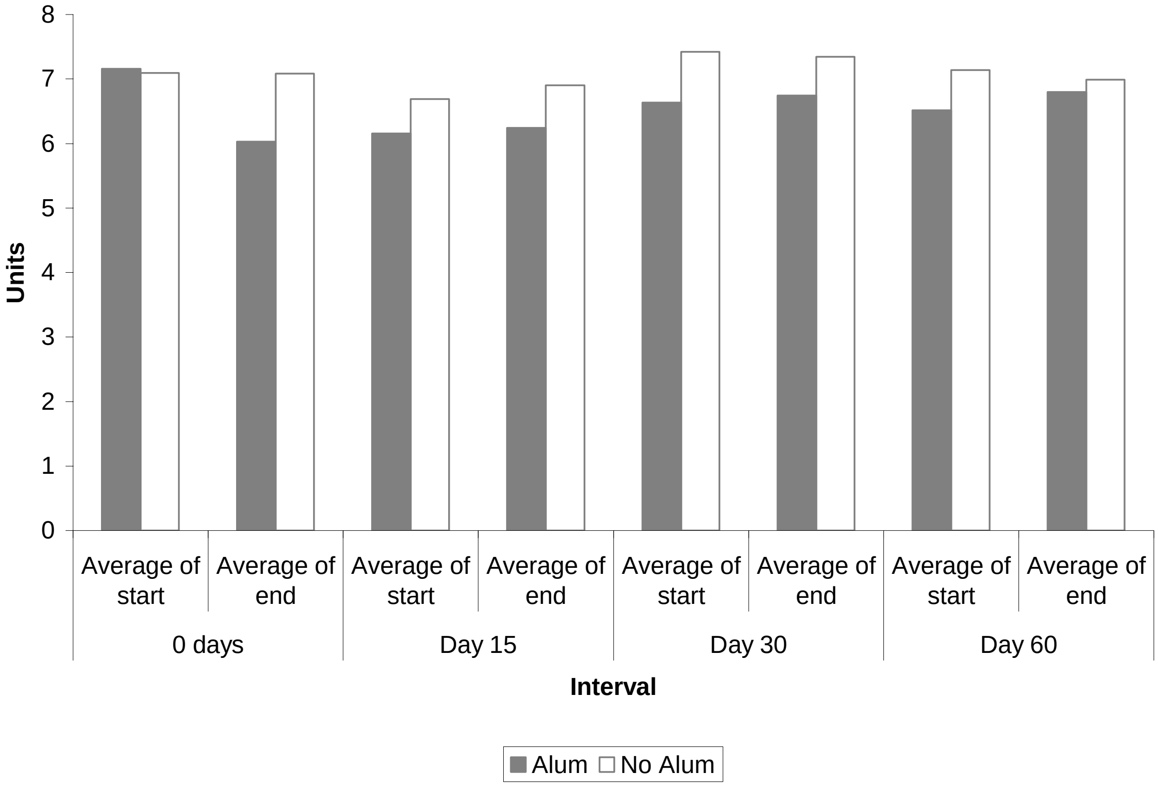


■	Alum
☐	No Alum

4.11: pH (suspended sediment concentration 300 g l^{-1})

Mean end pH was lower by (1.25 units) at the end of 0 Interval experiment compared to the start (Fig. 11) in Alum water. Mean pH levels for the rest of the intervals (i.e. Days 15, 30, and 60) in Alum water ended at higher mean levels than at the beginning of the experiments. By comparison, mean pH in no Alum water was lower at the end of the experiments at 0 Interval compared to the start whereas for Days 15, 30 and 60, mean pH was higher at the end of the experiments than at the beginning.

Figure 11: Mean pH in the Alum water column compared with mean pH in the no Alum water column pre- and post-experiment over four 15 day intervals (300 g l^{-1} conc)



4.12: Suspended Sediment Concentration (suspended sediment concentration 300 g l^{-1})

4.12.1: Zero Interval

Mean SS concentration of bottom waters from the Alum water column peaked at 0.008 g l^{-1} (time 30). Mean SS concentrations in the upper levels of the water column was variable with no obvious patterns emerging. In the no Alum water column, mean SS concentration bottom waters remained at consistent levels of ~0.004 g l^{-1} throughout the experiment while in the upper levels of the water column, mean concentrations showed variability in values (~0.003 g l^{-1}) at 0 Interval to near 0 g l^{-1} from ~time 30 (minimum turbidity).

4.12.2: Day 15

Mean SS concentration of bottom waters from the Alum water column peaked at 0 time at Day 15 (Figure 10 b). These waters show a reducing pattern of mean concentrations from 0.015 g l^{-1} (time 0) to 0.008 g l^{-1} by time 180. Mean SS concentrations in the no Alum water column peaked at 0.015 g l^{-1} (time 30) reducing to 0.005 g l^{-1} by time 180. In the upper levels of the water column, there is a reducing pattern of mean SS values for both Alum and no Alum treatments, reducing from a high 0.006 g l^{-1} (time 0) to a low 0.0025 g l^{-1} at time 180 where minimum turbidity occurred.

Figure 12 (a): Mean suspended sediment concentration in the Alum water column compared with mean suspended sediment in the no Alum water column at Zero Interval over time (300 g l^{-1} SS conc)

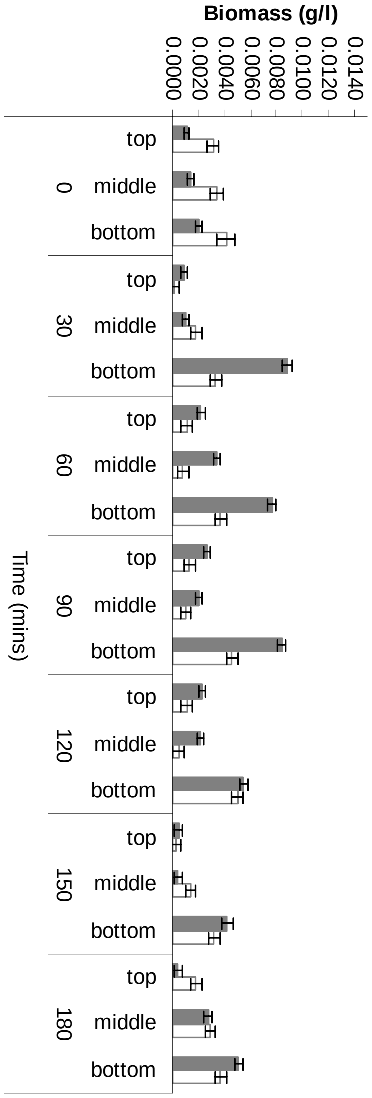
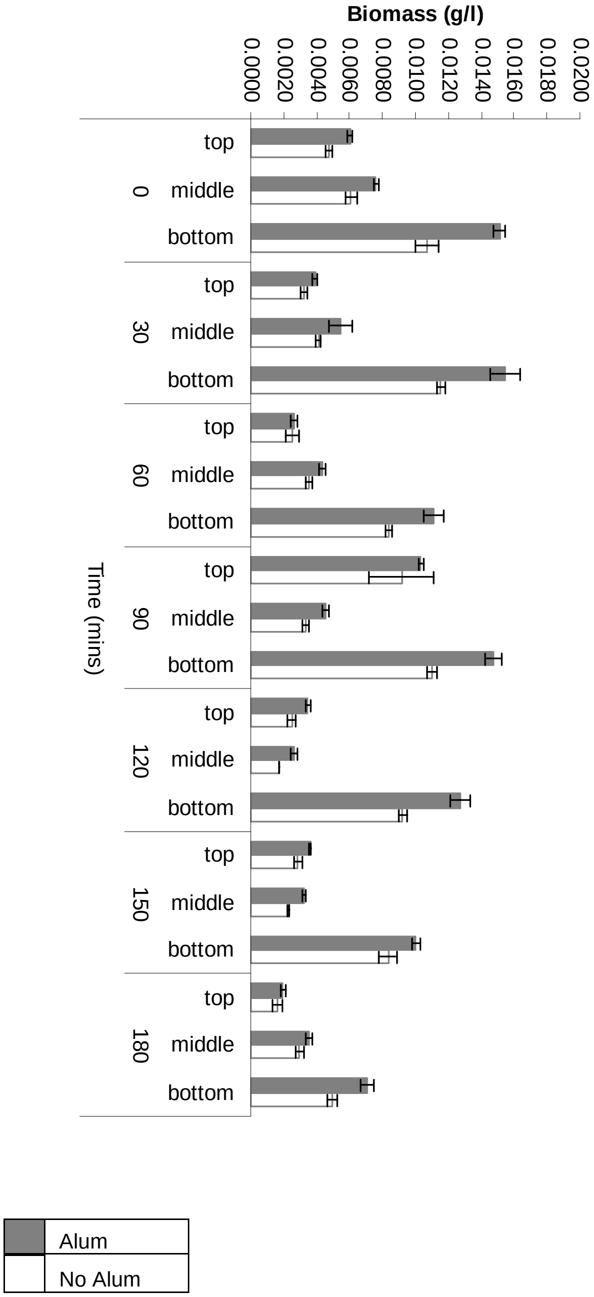


Figure 12 (b): Mean suspended sediment concentration down the Alum water column compared with mean suspended sediment down the no Alum water column at Day 15 over time (300 g l^{-1} SS conc)



4.12.3: Day 30

Mean SS concentration of bottom waters from the Alum water column were consistently higher than in the no Alum water column at Day 30 (Figure 10 c). Mean SS concentrations remained stable (at 0.002 g l^{-1}) in bottom waters with Alum and no Alum treatments over time. In the upper levels of the water columns, mean SS concentrations reduce from a high of ~ 0.0005 g l^{-1} (time 0) to ~50% of that value by time 150 with minimum turbidity occurring ~time 90.

4.12.4: Day 60

Mean SS concentration of bottom waters from the Alum water column at Day 60 (Figure 12 d), peaked at 0.005 g l^{-1} (time 60) then reduced to 0.0035 g l^{-1} by time 180. Mean SS concentrations of bottom waters from no Alum treatments peaked at 0.0058 g l^{-1} (time 150) with mean concentrations in the upper levels of the water column for both treatments, reducing from 0.0022 g l^{-1} (time 0) to 0.001 g l^{-1} by time 180.

Figure 12 (c): Mean suspended sediment concentration in the Alum water column compared with mean suspended sediment concentration in the no Alum water column at Day 30 over time (300 g l^{-1} SS conc)

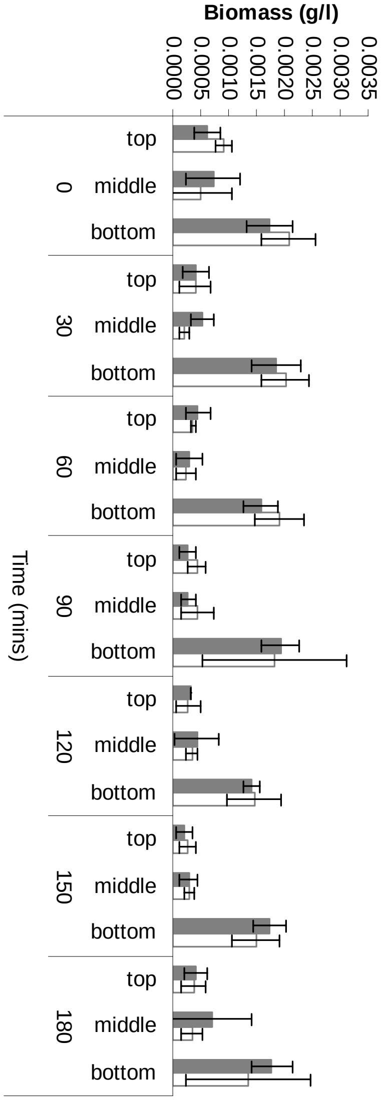
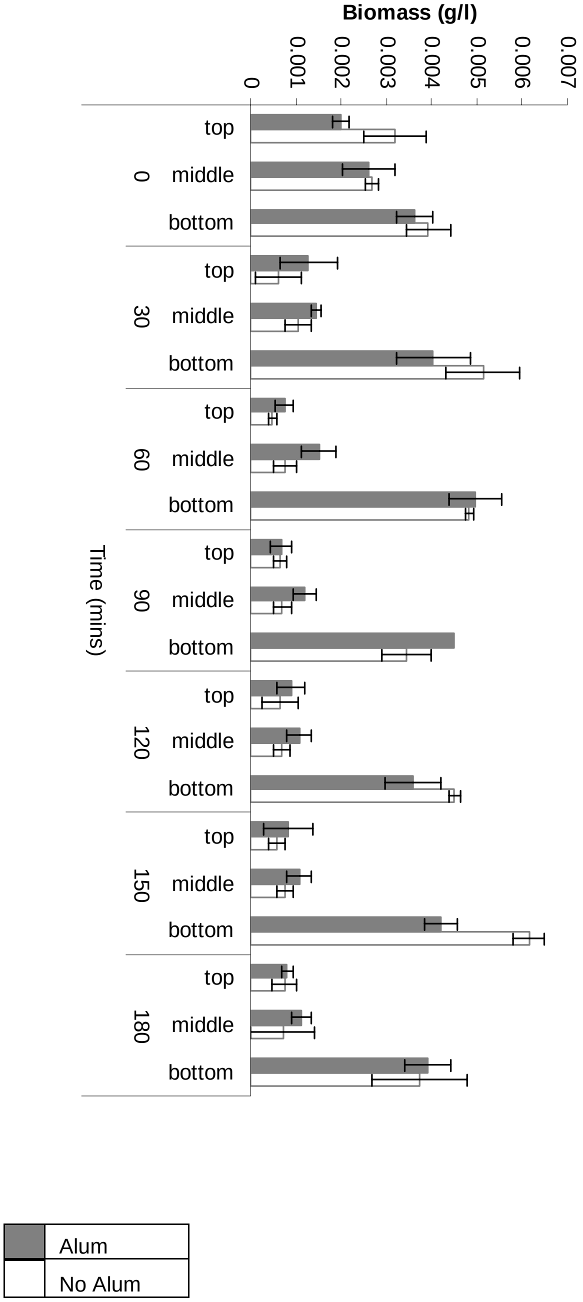


Figure 12 (d): Mean suspended sediment concentration in the Alum water column compared with mean suspended sediment concentration in the no Alum water column at Day 60 over time (300 g l^{-1} SS conc)



Statistically, there were two areas where Alum treatment was proven to be superior in settling sediment particles in comparison to no Alum. All three parameters exhibited significant responses to Alum treatment concurrently, firstly at Day 15, the point at which Alum treatment was shown to have the most significant and consistent response, and the other was at the 200 g l^{-1} SS concentration. A pattern of settling emerged at this point from the upper column where Alum treatment was having a significant effect, removing sediments faster across the intervals particularly at Day 15, than no Alum. This effect is corroborated by turbidity results (Figure 5 a – d) which shows a pattern of decreasing mean turbidity values signifying improved water clarity in the upper water column and the resultant increased turbidity in the bottom waters compared to no Alum treatment. At Day 15 the parameters were significant ($p<0.05$) at 100 g l^{-1} SS concentration and 300 g l^{-1} SS concentration. Mean suspended sediment levels at the 100 g l^{-1} SS concentration began erratically (0 Interval and Day 30) but stabilised at Day 15 and Day 60, where peak settling occurred toward the end of the settling period (time 180). At this lower concentration it was more difficult to detect differences evidenced by the wide error bars at Day 60. Variability was lower at the 200 g l^{-1} SS concentration than 100 g l^{-1} SS concentration and this variability had reduced further at the 300 g l^{-1} SS concentration.

At the 300 g l^{-1} SS concentration, the clearing effect from the upper column by Alum treatment was less obvious. At Day 15 the effect of Alum was inconsistent and there was little difference in the treatments although all parameters returned significant results at this point, and at Day 60. This observation is supported by significantly higher PAR levels ($p<0.05$) at the same time interval (Figures 4 b and d).

The parameters were least significant statistically at day 30 and day 60 (100 g l^{-1} SS concentration) where SS sediments remained significantly affected ($p<0.05$) by Alum treatment than without Alum. Suspended sediments was the parameter most affected by Alum treatment being featured significantly across all concentrations and time intervals.

Maximum water clarity (minimum turbidity) in the upper water column treated with Alum, occurred at varying time intervals across the experiments (see Appendix 1V). At 100 g l^{-1} SS concentration, the time to minimum turbidity averaged 85 minutes and at 200 g l^{-1} SS concentration, the time to minimum turbidity averaged ~115 minutes. By comparison, maximum water clarity did not occur at 100 g l^{-1} SS concentration or 200 g l^{-1} SS concentration within the 180 minutes allocated for the experiments with no alum treatment. At 300 g l^{-1} SS concentration, turbidity results showed minimum turbidity occurred at time 60 for Days 15, 30, and 60, PAR results show Days 15 and 30 averaged ~70 mins to minimum turbidity, and SS took an average 130 minutes to minimum turbidity.(0 Interval data for turbidity and PAR, and Day 60 SS, was irregular)

Similarities across the combinations were noticeable at the 100 g l^{-1} and 200 g l^{-1} SS concentration Time to minimum turbidity at 0 Interval (turbidity levels - 100 g l^{-1} SS concentration) was 120 minutes and this had increased to 150 minutes (PAR levels) and 150 minutes (SS levels). At Day 15, time to minimum turbidity was 60, 90 and 60 minutes respectively. At Day 30, times to minimum turbidity where 30, 60 and 30 minutes, and at Day 60, times were 120, 90 and 60 minutes. At 200 g l^{-1} SS concentration, the times to minimum turbidity at 0 Days were almost identical to the 100 g l^{-1} SS concentration. At Day 15, the times were almost identical but close to maximum time. Day 30 had an increasing pattern of times (from 60 minutes to 150 minutes) and Day 60 times were almost identical at 90 minutes. The 300 g l^{-1} SS concentration showed times at Day 15 ranging between 60 minutes and 150 minutes across the parameters and intervals but some of the data was inconclusive.

Turbidity of the top waters treated with Alum had reduced on average by over 76% at time 180 when compared to no Alum (see Appendix V1) at 200 g l^{-1} SS concentration. At the 300 g l^{-1} SS concentration, there was a difference in reduced turbidity of >52% in Alum compared to no Alum and at 100 g l^{-1} SS concentration the difference was >48% in Alum. Secchi levels were calculated using calculations from Kirk (1994) which showed an improvement of >40% (at 100 g l^{-1} SS concentration) in Alum compared to no Alum. Improvements in Secchi had occurred at the 200 g l^{-1} SS concentration and 300 g l^{-1} SS concentration at >17% and >19% respectively (see Appendix V11). A summary of parameters that resulted in significant change (derived from Mann Whitney *U* tests) is shown in Table 1

Table 1: Table comparing significance of results of settling experiment using Alum

100 g l^{-1} Suspended Sediment Concentration

	Zero	Day 15	Day 30	Day 60
S Sediments	<0.05	<0.05	<0.05	<0.05
Turbidity	<0.05	<0.05		
PAR		<0.05		
pH	<0.05			

200 g l^{-1} Suspended Sediment Concentration

	Zero	Day 15	Day 30	Day 60
S Sediments	<0.05	<0.05	<0.05	<0.05
Turbidity	<0.05	<0.05		<0.05
PAR	<0.05		<0.05	<0.05
pH				

300 g l^{-1} Suspended Sediment Concentration

	Zero	Day 15	Day 30	Day 60
S Sediments	<0.05	<0.05	<0.05	<0.05
Turbidity	<0.05	<0.05		
PAR		<0.05	<0.05	<0.05
pH	<0.05	<0.05	<0.05	<0.05

4.13: Discussion

It has long been recognized that Alum (aluminium sulfate) can be used for coagulating organic and inorganic particles to help clarify water (e.g., Tucker, 1892). Alum increased the settling rate of Lake Waikare sediment particles, and improved water clarity, relative to that of non-treated water in my laboratory study. This was evident from generally decreased turbidity and suspended sediment levels at the surface of the settling chambers, and increased levels in bottom samples from the water column treated with Alum compared with no Alum, and the increased irradiance levels (PAR) in Alum treated replicates at identical time intervals. Whole lake studies conducted by Environment Bay of Plenty (EBOP) at Lake Okaro, North Island, New Zealand, have also indicated increased sediment particle settling occurred during and after Alum additions (Paul and Hamilton, 2003). Quinn *et al.* (2004) conducted a laboratory study using two Alum dose rates to test the effect of Alum on Lake Okaro water and found turbidity was reduced markedly with Alum treatment compared to control experiments. In Horseshoe Lake (Wisconsin USA), Secchi disc transparency increased after the application of Alum and similarly, in Medical Lake (Washington, USA) an increase in water transparency was observed after Alum was applied (Cooke and Kennedy, 1981).

Other studies on North American lakes have showed increased Secchi depths ranging between 13% and 87% have been achieved with a single dose (Welch and Shrieve, 1994). Cooke and Kennedy (1981) in a survey of 28 lakes (including reservoirs and ponds) treated with Alum in the 1970's, obtained mixed results from the treatment of Alum as it applied to water clarity and all but one was treated with a single dose. My analysis of equivalent Secchi depth levels showed improvements with Alum averaged 18% (100 g l^{-1} SS concentration), 15% (200 g l^{-1} SS concentration) and 12% (300 SS concentration) compared to no Alum. Improvements in turbidity with Alum averaged 25% (100 g l^{-1} SS concentration), 40% (200 g l^{-1} SS concentration) and 23% (300 SS concentration) compared to no Alum. Quinn *et al.* (2004) report turbidity levels improved by between 40% and 80% with Alum treatment compared to without Alum. Wang *et al.* (2002) used jar tests to remove organic matter and dissolved organic compounds (DOC) at Alum dose rates of between 20 mg l^{-1} and 120 mg l^{-1} (pH 6). They achieved 50% removal of organic matter at dose rates of 70 mg l^{-1} – 110 mg l^{-1} (pH 6) and 80% removal of DOC at 80 mg l^{-1} (pH 8). Ackers and Haddad (1987) used 3 mg l^{-1} to cause the complete removal of turbidity from moderately turbid water (100 Formazin Turbidity Units, FTU = ~NTU) in a conventional jar test.

The use of Alum is a widely applied lake restoration measure for the precipitation and inactivation of P (Lewandowski *et al.,* 2003). In lakes where Alum was used to inactivate/precipitate P, the dose rates averaged 0.007 mg l^{-1} (Cooke *et al.,* 1981). Quinn *et al.* (2004) used dose rates of between 0.500 mg l^{-1} to 40 mg l^{-1} that resulted in turbidity levels being between 40% and 80% of control levels. Welch and Shrieve (1994) used between 0.005 mg l^{-1} and 0.0110 mg l^{-1} Alum to dose a range of lakes in the USA. Alum treatment of polymictic lakes has been successful in reducing the photic zone and longevity of the treatment has been in the 4 – 9 year range (Cooke *et al.,* 1993).

Alum treatment was consistently effective in settling sediment particles for up to 15 days at two of the three concentrations I tested, i.e. at 100 g l^{-1} and 200 g l^{-1}. The responses from turbidity testing, PAR and suspended sediment concentrations concurrently, indicated sediment particles settled significantly faster in Alum treated water than without Alum. The average time to achieve maximum water clarity in the Alum treated water column (100 g l^{-1}.SS concentration) was 85 minutes and at 200 g l^{-1}.SS concentration, the average time had increased to ~115 minutes. Zero Interval, Day 15 and Day 30 achieved clear water clarity at times that were similar across the parameters. Across the concentrations, there is a pattern of increased times with increased concentrations (e.g. Day 15 and Day 30) (see Appendix 1V).

The effects of improved settling from Alum treatment beyond 15 days (Day 30) remain positive with significantly greater responses ($p<0.05$) from SS concentration than no Alum treatment across the concentrations and time intervals. At 60 days, SS concentration was still responding to Alum treatment at significantly higher ($p<0.05$) levels than no Alum while at the 200 g l^{-1} SS concentrations PAR and turbidity was significantly higher ($p<0.05$) than no Alum and settling times to minimum turbidity, were still occurring within time 150 where no Alum treatment did not settle (within time 180).

Mean pH at the conclusion of the experiments gradually increased over time. Mean pH levels at 300 g l^{-1} SS concentration were significantly higher at the end ($p<0.05$) than at the beginning and this was also true at 100 g l^{-1} SS concentration (0 interval). From these results, it would appear that Alum treatments were becoming less acidic through time indicating a chemical change occurring in the replicates over time. Lake water contains bicarbonate of lime in solution and at higher pH concentrations the water becomes less acidic (alkaline) (Tucker, 1892b). When Alum is added to water it dissociates, aluminium ions appear and aluminium hydroxide forms of $Al(OH)_3$ and a decrease in pH occurs as   When Alum is added to soft water (or if water becomes acidic), its acid-neutralising capacity quickly diminishes and soluble aluminium species dominate e.g. toxic aluminium forms of $Al(OH)_2$+ and Al^3 (Cooke et al., 1993). I did not analyse the reactions chemically but suspect precipitation of HCO_3 was occurring. The lowest recorded pH was 5.5 (0 Interval: 300 concentration) but the trend is evident and in time the hydrated and soluble Al^{3+} will dominate if levels decline drastically. The implication then is that the threat of toxicity should the trend continue, is a strong possibility.

The Alum dose used in my study was 35 mg l^{-1}, comparable with rates used by New Zealand's largest metropolitan city to treat domestic water supplies at maximum turbidity 40 NTU. This dose rate was derived from methods and procedures of Cooke and Kennedy (1981); to do so, a titration of a known concentration of alum was added to replicates of lake water until pH 6 was reached. Aluminium solubility is pH-dependant with maximum floc formation occurring at between pH 6 – 8 (Cooke *et al.,* 1981). The concentration I used was therefore the maximum possible for Lake Waikare.

Turbidity was always higher in Alum treatments than without Alum at similar time intervals. Alum (Aluminium Sulphate) flocculates (binds) sediments including particulate and organic matter in the water column together. The particles increase in size and settle out of the water column more quickly than without Alum. Alum as a flocculent is used widely, for example, most water treatment plants in New Zealand (and elsewhere) use Alum to settle sediment particles in water for domestic use (Ministry of Health, 2005). Jar tests employed by the wastewater industry use dose rates of between 0.015 g l^{-1} and 0.030 g l^{-1} (Ministry of Health, 2005). A laboratory study of laundry waste water treatment conducted by the US Army Facilities Engineering Support Agency used Alum and their study showed turbidity had improved from 400 FTU to mean 5 FTU (n=3). They concluded Alum achieved good flocculation (and reduced turbidity) in the range 5.5 – 6 pH and 25–35^{o}C (Botros and Best, 1977). In industry, the treatment of dairy factory waste-water by coagulation and decantation uses Alum to obtain optimal removal of suspended matter (94% of suspended solids, 89% of total phosphorus). This process uses a dose rate of between 8 x 10^{-7} g l^{-1} – 0.001 g l^{-1} (Hamdani *et al.,* 2005). In pulp mill waste water treatment, Alum is used for the coagulation-flocculation of Chemical Oxygen Demand (67%) and colour removal (98%) but at significantly higher levels of between 350 g l^{-1} and 400 g l^{-1} Alum (Buzzini *et al.,* 2005)

Chemical coagulation with Alum followed by filtration is another common method of treating water for domestic use in New Zealand (Ministry of Health, 2005). This combined process has proved to be effective in removing dissolved and colloidal colour, turbidity, algae and other suspensoids (bacteria, viruses protozoa etc)

(Ministry of Health, 2005). These particles form a floc during coagulation in a chamber containing a contact medium, trapping the particles until floc build-up clogs the media which is then removed by back-washing. Where hydroxide precipitation is important for "sweep flocculation", pre-hydrolyzed coagulants, such as polyaluminium chloride (PACl) are widely used.

Sweep flocculation occurs when impure particles enmeshed in the growing floc are removed by sedimentation, swept out of the water column by the amorphous hydroxide precipitate (Gregory and Duan, 2001) formed by the addition of Alum (Welch and Shrieve, 1998). Hydrolyzing coagulants such as polyaluminium chloride are extensively used in water and wastewater treatment and have several advantages over traditional additives such as Alum (Gregory and Dupont, 2001). Coagulant floc breakage appears to be essentially irreversible and an advantage of polyaluminium chloride products is that they form larger and stronger flocs than Alum although sediment volumes are slightly lower. However, the effective and consistent removal of organic material (total organic carbon) contained in the raw water was not achieved by adsorption (powdered activated carbon) or coagulation (Berube *et al.,* 2002).

Chitosan is an acetylate of chitin, a biodegradable, cationic polymer. Chitosan was assessed as an on-site treatment agent of turbid water caused during river construction works. Jar tests have indicated that floc of Chitosan is much larger than that of aluminium sulfate and turbidity under moving water conditions, is much lower than that of aluminium sulfate. Chitosan was applied where river construction work was being undertaken and at two other sampling stations stretching 600 m downstream where turbidity was monitored. Turbidity at each station had drastically reduced indicating that Chitosan could be helpful in mitigating problems emanating from turbidity-causing activities in or near moving water (Sekine *et al.,* 2006). These examples of the use of flocculants are alternative methods that may have applications in lake restoration. While there is potential for chemical treatments, filtration options will be limited.

4.14: Conclusions

Alum is used widely as a flocculent (Welch and Cooke, 1999) and I found flocculation with Alum was effective up to 15 days in laboratory jar tests. Alum treatments were consistently superior in settling suspended sediment particles compared to no Alum treatments and results from three parameters monitored were significant concurrently, at two of 12 possible combinations I tested, and one parameter exhibited significant responses consistently across the combinations.

Significant chemical changes occurred in replicates treated with Alum identified by changes in pH that should the trend identified continue, toxicity to biota is likely to occur. Additional follow-up doses of Alum therefore must be considered carefully before proceeding. While a doubling of my dose will still be ~50% of the combined dose reported by Cooke and Kennedy (1981), the extent and level of toxicity is unknown but the potential threat of toxicity occurring in Lake Waikare increases with increased levels of Alum.

Alum succeeded in removing suspended sediments from the water column at rates faster than in no Alum, Where results were conclusive, Alum treated water had reached minimum turbidity (high clarity) within the 180 minute time frame allocated but no alum water did not. Alum treatment responses were detected to Day 60 at the highest SS concentration indicating Alum treated water was still active after 60 days. My analysis of mean Secchi depth levels showed similar improvements (~15%) to those achieved by Welch and Shrieve (1994) with a single dose of Alum. Turbidity had improved by ~30% and at Day 60, was still achieving 12% improvements with Alum compared to no Alum.

Chapter 4

4. Summary Conclusions

The growth performance of *C. corallina* plantlets placed in exclosures in Lake Waikare was expected. The plantlets grew where the influence of fish herbivory and disturbance was inhibited and where the wave climate would be the most subdued, at the sheltered site. What was not expected was growth occurred where fish herbivory was uninhibited. Charophytes grew in the presence of fish in winter at the sheltered site but at growth rates less than those inside exclosures
The variation in the growth rates between outside the exclosures and inside suggests fish predation caused this variation because fish are widely known to disturb and graze submerged macrophytes (Reynolds, 1994). The current fish population in Lake Waikare can inflict direct damage on submerged macrophytes but a slowed metabolism in winter could account for the reduced grazing on exposed plantlets in winter. Good Water depth between the sites differed by ~100 mm and overall maximum depth was ~1 m indicating fish feeding behaviour should not be affected greatly by water depth to the same extent as in stratified lakes. In summer, the plants were completely damaged so this is the site that fish herbivory requires the most control to allow charophytes to survive and grow.

Current fish densities for Koi carp (220 kg ha^{-1}) (B. Hicks, Waikato University, pers comm.) means there are potentially 750 tonne of Koi in the lake and removing this biomass of fish could be the catalyst the lake requires to ensure successful re-establishment of charophytes. Stephens *et al*. (2004) in their assessment of Lake Waikare, and de Winton (2002) in Lake Rotoroa, state coarse fish are a threat to submerged vegetation establishment and to improve the light climate to a level required to establish aquatic plants, would require fish removal, a reduced wave climate, a reduction in water levels (to improve light penetration) and a reduction in nutrient levels.

Five weeks was the minimum for charophyte plantlets to attach their root system, recover from transplanting and exhibit signs of growth in the laboratory. Plantlets grew at higher rates at the sheltered site in winter where suspended sediment concentration was lower (by 21%) than at the exposed site. Further, wave exposure was higher at the exposed than the sheltered site in winter and summer. Therefore, establishing submerged macrophytes in the lake will be most successful at this site. Exclosures will be required firstly, to inhibit fish herbivory and because plantlet growth rates increased 87% inside exclosures but only 52% outside exclosures. Dugdale *et al.* (2005) and de Winton et al. (2002) had success with re-establishing charophytes in Lake Rotoroa using exclosures and potted plants and I had success growing charophytes in a sheltered embayment Reeves *et al.* (2002) had suggested, as well as at an exposed site. Wave baffles they promote and the knowledge charophytes can be established in the absence of fish (this study) means the positive feed back they expected i.e. dampening further the wave orbital velocities and consolidating lake sediments, will lead to improving the light climate and enhance submerged vegetation growth (Reeves *et al.,* 2002).

Charophytes dry weight biomass was greater (by ~40%) at the sheltered than the exposed site by the experiments termination in winter. In summer, however, dry weight biomass was greater (by 21%) at the exposed site. A better light climate was likely the main factor contributing to improved charophyte growth rates at the exposed site in summer compared to the sheltered site although the euphotic depth I calculated showed consistent levels at both sites. In winter, the euphotic depth was greater at the sheltered site where greater growth rates were achieved.

The light climate in Lake Waikare is affected by heavy sedimentation, high algal biomass, and high suspended sediment concentrations. To potentially improve the light climate I used Alum in a range of experiments to test if suspended sediments in Lake Waikare water could be settled out long enough to allow germination an/or plant growth to occur. I found Alum was effective for at least 15 days at suspended sediment concentration 200 g l^{-1} but it is unclear if these results alone are sufficient evidence Alum will improve the light climate in Lake Waikare of sufficient duration to allow charophytes to establish and grow. While Alum remained reasonably active to Day 60 at the higher concentration of 300 g l^{-1} suspended sediment concentration, its response time to clearing water had increased to 150 minutes (from 60 minutes) and if it remained active through another 15 day interval, charophyte plantlets could be established because it took ~30 days to establish them in controlled conditions in the laboratory.

Oospores from Lake Rotoroa took five weeks to germinate in an incubator in the laboratory under ideal conditions, but 90 days was insufficient germination time in Lake Waikare. Reeves *et al.* (2002) identified an absence of a viable propagule seed bank in Lake Waikare and the sporadic heavy sediment loading they reported, and observed in my study, were the major causes of germination failure of the oospores. The heavy sedimentation encountered highlights the enormity of the challenge facing lake managers trying to re-establish submerged macrophytes from naturally occurring oospores where they are known to exist. But in lakes like Lake Waikare where oospores are known to be sparse, the problems are compounded. Oospores are a vital component in the re-establishment of submerged macrophytes because sparse propagules in lake sediments, and light availability are obstacles to plant recovery (de Winton, *et al.,* 2002). Therefore they will need to be re-introduced.

My study shows the removal of fish combined with a reduction of sedimentation could allow survival of charophytes, but that establishment of new plantlets is currently impossible based on sedimentation alone. Introduction of new oospores (sensu Reeves *et al.,* 2002) to Lake Waikare, which has a depleted seed bank, would therefore not be adequate. Although fish could be removed or prevented from accessing charophytes in plots (e.g., de Winton *et al.,* 2002; Dugdale *et al.,* 2005), which is particularly needed in summer when I observed all charophyte biomass to be removed, reducing the suspended sediment loads in the water column may not be achieved to an adequate degree by Alum treatment alone. The potential therefore exists for using alternate chemicals (e.g. Chitosan) either in isolation or in conjunction with Alum to mitigate any potential toxicity to biota that may occur.

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Appendices

Appendix 1: Oospore Concentration

C. corallina Oospore concentration determined from sub-sampling 300 g oospore-laden Lake Rotoroa sediment (n=3)

Replicate	n	Vol mls	Oospore ml^{-1}	Total	Mean	SEM
1	10	50	1.22	61	6.1	1.057
2	10	25	6.16	154	15.4	1.299
3	10	35	0.914	32	3.2	0.628

(combined mean 8.233; combined SEM 3.121)

Appendix 11: Lake Waikare Wet Sediment Weight Determination

Sediment Wet Weight required to achieve suspended sediment concentrations

Sediment Concentration (g m^{-3})	Wet Sediment Weight (g)
300	9.974
200	6.51
100	3.254

Appendix 111: Random Order of Settling Experiments

SRS order determined for replicates

Order	Replicate	Order	Replicate	Order	Replicate	Order	Replicate
1	24	**7**	1	**13**	19	**19**	13
2	6	**8**	17	**14**	21	**20**	4
3	3	**9**	14	**15**	7	**21**	8
4	23	**10**	9	**16**	10	**22**	15
5	18	**11**	2	**17**	5	**23**	11
6	22	**12**	12	**18**	16	**24**	20

Appendix 1V: Euphotic depths in summer and winter [calculated from Kirk, 1994: E_0 (Z_{eu}) = $0.01E_o$(O-) where E_o denotes PAR; O- denotes null depth (sub-surface)]

Winter

Date		Inside	Outside
4th Sept	exposed	1.45	1.60
	sheltered	1.97	1.78
11th Sept	exposed	0.85	0.76
	sheltered	0.99	0.85
18th Sept	exposed	1.34	0.73
	sheltered	1.89	1.47
21st Sept	exposed	0.96	1.47
	sheltered	1.10	1.07

Summer

Date		Inside	Outside
8-Dec	exposed	1.29	1.15
	sheltered	1.19	1.35
15-Dec	exposed	0.54	0.77
	sheltered	0.66	0.64
22-Jan	exposed	0.60	0.01
	sheltered	1.15	0.01
29-Jan	exposed	0.65	0.64
	sheltered	0.84	0.80
5-Jan	exposed	0.45	0.46
	sheltered	0.64	0.58
19-Jan	exposed	0.54	0.52
	sheltered	0.74	0.76
26-Jan	exposed	0.51	0.69
	sheltered	0.39	0.39
9-Mar	exposed	0.53	0.55
	sheltered	0.65	0.63

Appendix V: Time (mins) to maximum water clarity (minimum turbidity) in Alum-treated top waters (SS concentration 100 g l^{-1} and 200 g l^{-1}).

Turbidity					PAR				SS			
SS Conc	**0 days**	**15 days**	**30 days**	**60 days**	**0 days**	**15 days**	**30 days**	**60 days**	**0 days**	**15 days**	**30 days**	**60 days**
100	120	60	30	120	150	90	60	90	150	60	30	60
200	120	150	60	90	150	180	90	90	90	150	150	60
300	IC	60	60	60	IC	120	90	IC	150	150	90	IC

(IC: inconclusive)

Appendix V1: Turbidity comparison in "top" waters of settling water column (at time 180) between Alum treatment and no Alum treatment

100 g l^{-1} SS concentration

Interval	No Alum	Alum	%age change
0	34.1	17.77	+ 48
15	18.5	17.92	+ 3
30	10.3	15.64	- 34
60	13.3	15.28	- 13

200 g l^{-1} SS concentration

Interval	No Alum	Alum	%age change
0	22.1	5.15	+ 76
15	17.1	9	+ 47
30	12.8	9.86	+ 23
60	12.5	11	+ 12

300 g l^{-1} SS concentration

Interval	No Alum	Alum	%age change
0	26.3	12.5	+ 52
15	9.6	8.6	+ 10
30	10.1	8.2	+ 18
60	9.3	8.2	+ 12

V11: Secchi Depths calculated from Kirk, (1994), SD = 2.2/Kd)

100 g l^{-1} SS Concentration

Interval	No Alum	Alum	%change
0	0.78	0.80	+ 15.21
15	0.73	1.20	+ 40.39
30	0.83	0.90	+ 7.22
60	0.66	0.75	+ 12.12

200 g l^{-1} SS Concentration

Interval	No Alum	Alum	%change
0	0.50	0.60	+ 17.20
15	0.60	0.56	- 7.1
30	0.52	0.59	+ 13.23
60	0.55	0.66	+ 15.77

300 g l^{-1} SS Concentration

Interval	No Alum	Alum	%change
0	0.43	0.46	+ 6.48
15	0.40	0.42	+ 5.15
30	0.42	0.52	+ 18.99
60	0.44	0.54	+ 18.78