Review of Best Management Practices for Aquatic Vegetation Control in Stormwater Ponds, Wetlands, and Lakes

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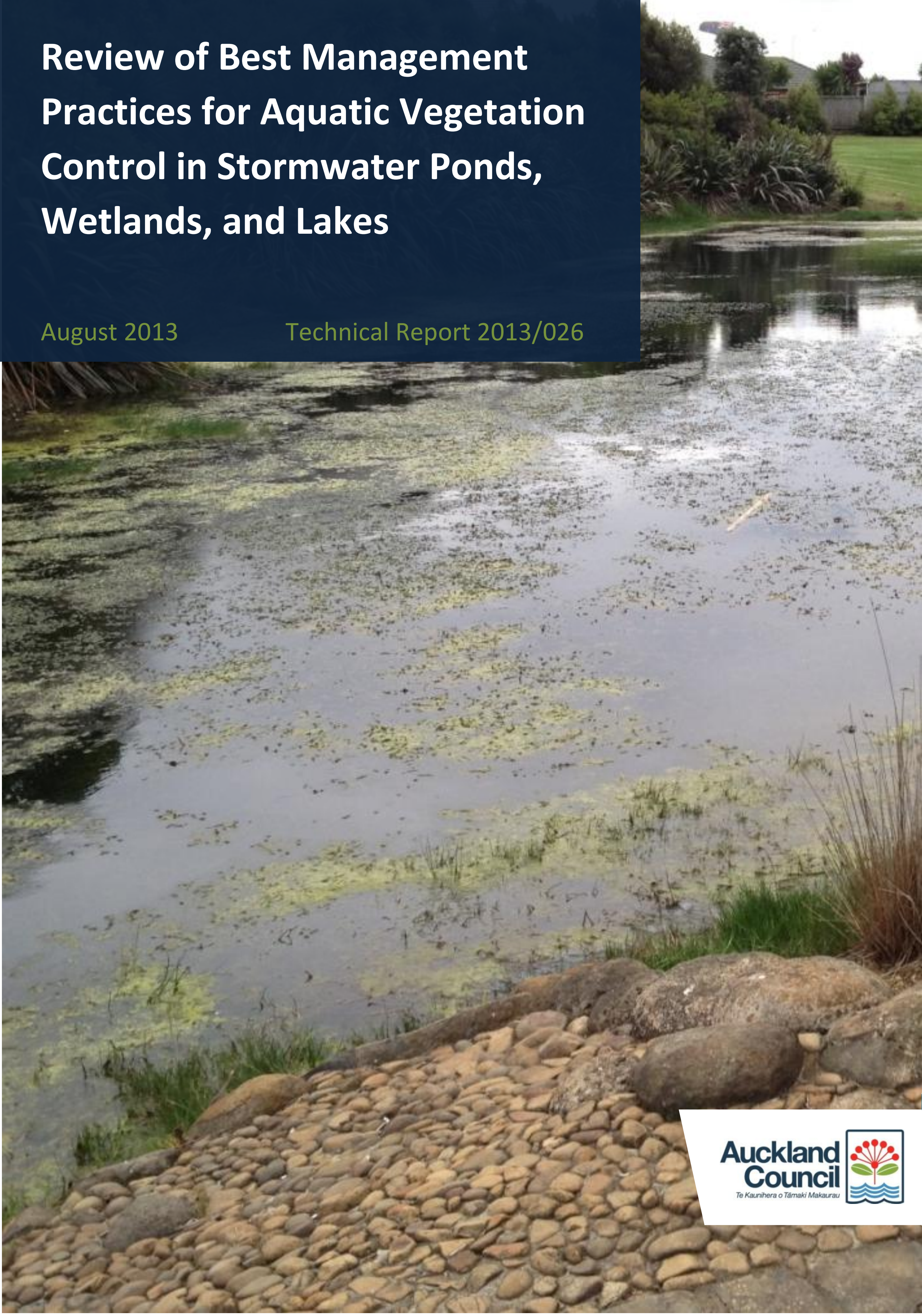

Auckland Council

Te Kaunihera o Tāmaki Makaurau

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Review of best management practices for aquatic vegetation control in stormwater ponds, wetlands, and lakes

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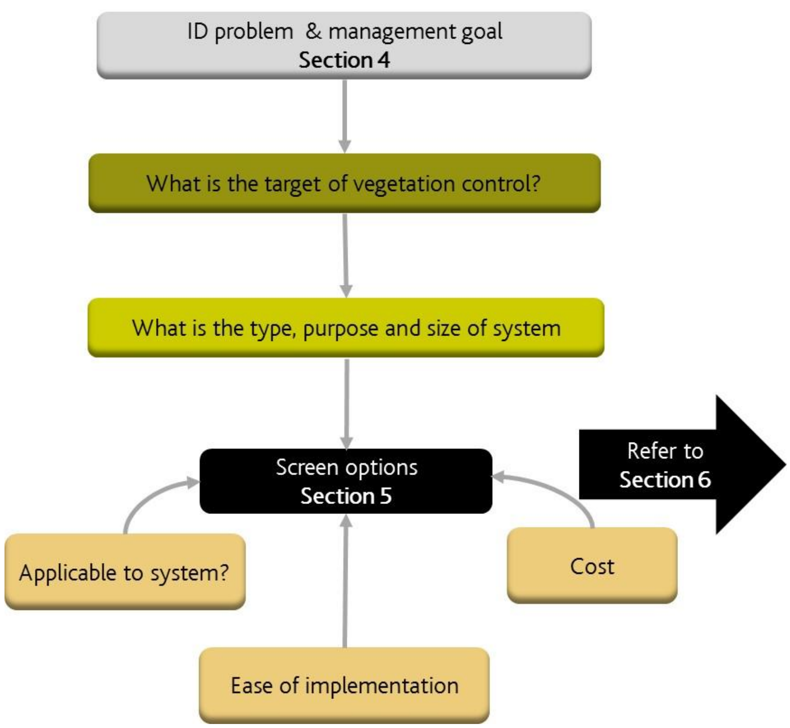
Executive Summary

Auckland Council (AC) is responsible for the development and operation of a stormwater network across the region to avert risks to citizens and the environment. Within this stormwater network, aquatic vegetation (including plants, unicellular and filamentous algae) can have both a positive and negative role in stormwater management and water quality treatment. The situations where management is needed to control aquatic vegetation are not always clear, and an inability to identify effective, feasible and economical control options may constrain management initiatives.

AC (Infrastructure and Technical Services, Stormwater) commissioned this technical report to provide information for decision-making on aquatic vegetation management within stormwater systems that are likely to experience vegetation-related issues. Information was collated from a comprehensive literature review, augmented by knowledge held by the authors. This review identified a wide range of management practices that could be potentially employed. It also demonstrated complexities and uncertainties relating to these options that makes the identification of a best management practice difficult. Hence, the focus of this report was to enable users to screen for potential options, and use reference material provided on each option to confirm the best practice to employ for each situation.



The report identifies factors to define whether there is an aquatic vegetation problem (Section 3.0), and emphasises the need for agreed management goals for control (e.g. reduction, mitigation, containment, eradication). Resources to screen which management option(s) to employ are provided (Section 4.0), relating to the target aquatic vegetation, likely applicability of options to the system being managed, indicative cost, and ease of implementation. Initial screening allows users to shortlist potential control options for further reference (Section 5.0).

Thirty-five control options are described (Section 5.0) in sufficient detail to consider applicability to individual sites and species. These options are grouped under categories of biological, chemical or physical control. Biological control options involve the use of organisms to predate, infect or control vegetation growth (e.g. classical biological control) or manipulate conditions to control algal growth (e.g. pest fish removal, microbial products). Chemical control options involve the use of pesticides and chemicals (e.g. glyphosate, diquat), or the use of flocculants and nutrient inactivation products that are used to reduce nutrient loading, thereby decreasing algal growth. Physical control options involve removing vegetation or algal biomass (e.g. mechanical or manual harvesting), or setting up barriers to their growth (e.g. shading, bottom lining, sediment capping).

Preventative management options are usually the most cost effective, and these are also briefly described (Section 6.0). For example, the use of hygiene or quarantine protocols can reduce weed introductions or spread. Catchment-based practices to reduce sediment and nutrient sources to stormwater are likely to assist in the avoidance of algal and possibly aquatic plant problems. Nutrient removal may be a co-benefit where harvesting of submerged weed biomass is undertaken in stormwater systems. It should also be considered that removal of substantial amounts of submerged vegetation may result in a sudden and difficult-to-reverse switch to a turbid, phytoplankton dominated state. Another possible solution is the conversion of systems that experience aquatic vegetation issues, to systems that are less likely to experience issues.

The focus of this report is on systems that receive significant stormwater inputs, i.e. constructed bodies, including ponds, amenity lakes, wetlands, and highly-modified receiving bodies. However, some information will have application to other natural water bodies.

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Glossary

Terms shown in bold in the text are included in the following glossary.

AEE (Assessment of Environmental Effects): An assessment of the possible positive or negative impact that a proposed project may have on the environment, together consisting of the environmental, social and economic aspects.

a.i.: Active ingredient, the component responsible for a herbicides ability to control the target pest.

Amenity control: Vegetation removal to allow intended use of facilities or enable access.

Anoxic: greatly depleted in oxygen.

Arthropods: invertebrate having jointed limbs, a segmented body and a hard exoskeleton.

Avian botulism: A common disease of waterfowl that results when birds ingest toxin produced by bacteria, with outbreaks associated with temperatures over 25°C, low dissolved oxygen, water pH of 8-9, salinity of 2-4, and presence of decaying organic matter.

Biocontrol: Use of organisms that are natural predators, parasites, or pathogens of a pest.

Clam shell bucket: A hinged digger bucket forming a claw-like structure.

Chlorophyll *a*: A form of green pigment found in plants and photosynthetic algae or bacteria that traps the energy of sunlight for photosynthesis.

Coarse fish: Species targeted for angling that are not salmonid sports fish, e.g. tench, rudd, koi carp, perch.

Cyanobacteria: A division of microorganisms related to the bacteria but capable of photosynthesis.

Detritus: Non-living particulate organic material.

Diadromous: Fish migrating, or moving for breeding, between fresh and salt water.

Efficacy: The ability to produce a desired or intended result.

Emergent plant: With foliage above the water level.

Emetic: A substance that causes vomiting.

Fork length: A measure of fish size from the nose to the deepest part of the tail fork.

Half maximal effective concentration EC_{50}: The concentration of a compound where 50% of the test organism population exhibit a response after the specified exposure duration.

Harmful algal blooms (HABs): Blooms of phytoplankton known to naturally produce biotoxins.

Herbivorous: Predominantly plant eating.

Isolates: A pure strain separated from a fungal culture.

Mesocosm: An enclosure within which an experiment can be performed in close to natural conditions.

Naturalise: Establish self-sustaining populations in the wild.

Not adequately controlled: 2-3 treatments per season is not sufficient to kill plants.

Operational Plan: A written agreement outlining the objective of grass carp stocking, identifying the site, security measures and monitoring techniques, outlining introduction and recovery procedures and conditions of the approval.

Phytoplankton: Microscopic plants.

Phytoremediation: The use of plants to remove or neutralize contaminants, as in polluted soil or water.

Planktivorous: An animal feeding primarily on plankton.

Rhizomes: A horizontal, usually underground stem that often sends out roots and shoots from its nodes.

Sporulate: Produce or form a spore or spores.

Stenching agent: An unpleasant odour to alert handlers of chemical substances.

Stratification: The separation of lakes into three layers, usually driven by heating of surface waters (the epilimnion), which are then separated from colder bottom waters (the hypolimnion) by a middle layer that exhibits a strong gradient in temperature (called the metalimnion or thermocline).

Total control: Removal of most to all submerged vegetation.

Trammel net: A tri-layer of netting of two mesh sizes and tensions.

Virulence: The degree of pathogenicity of a microorganism.

Zooplankton: Small animals and the larvae of larger animals.

1.0 Introduction

Auckland Council (AC) is responsible for the development and operation of a stormwater network across the region to protect ratepayers' properties against flooding and avert risks to citizens and the environment. Additional management objectives have arisen from the recognition of the role of the stormwater network in providing ecosystem services and amenity values for the area.

Aquatic vegetation, including filamentous and unicellular algae, play a role as primary producers in aquatic ecosystems, even in the highly modified systems of urban areas. They can contribute to the processing or retention of nutrients and contaminants, retain water through amending flows, and provide habitat or food for biota. However, in some cases their development is at odds with the management purposes of stormwater systems and options are sought to reduce or remove their influence.

The Aquatic Plant Management Society (USA) define control as: 'techniques used alone or in combination that result in a timely, consistent, and substantial reduction of a target plant population to levels that alleviate an existing or potential impairment to the uses or functions of the water body'. This definition allows for a range of outcomes that might include weed eradication, suppression or containment, or some level of mitigation for an impact. The goals of control should always be clearly identified.

The situations where aquatic vegetation requires active management are not always clear and depend upon ecological, economic, and social considerations as well as the availability of effective and feasible options to control, eradicate, or mitigate aquatic plant or algal problems.

This technical report provides information for decision-making on the management of aquatic vegetation within the stormwater network of the Auckland region, including guidance on which situations require the instigation of aquatic vegetation control, and which management option(s) to employ. Use of these resources and decision making tools may guide operational activities through to planning levels within Council.

This report has four main sections:

- An outline of methods used to discover and screen relevant information from the literature.
- Steps required identifying situations requiring aquatic vegetation control.

- A framework to screen possible options for aquatic plant and algae management.
- Descriptions of control options that may be applicable to the stormwater network of the Auckland region.

The focus of this report is on systems that receive significant stormwater inputs, i.e. constructed bodies, including ponds, lakes, wetlands, and highly-modified receiving bodies. Only systems that hold water for a considerable period of time (>2-4 months) are considered, as systems such as rain gardens, flood detention basins and swales unlikely to experience aquatic vegetation issues. However, some information will have application to other natural water bodies.

2.0 Literature review

2.1 Methods

A literature search was made of major online reference and full-text databases (Web of Science, ASFA, NIWA library catalogue, Google Scholar) using specific search terms to extract relevant references for the time period since the year 2000. Search terms included:

- Group 1: stormwater device, detention basin/pond, constructed wetland, lake, sediment forebay.
- Group 2: vegetation, macrophytes, plants, weeds, algae, harmful algae bloom (HAB).
- Group 3; control, mitigation, management, containment, eradication, restoration, rehabilitation.

In addition, relevant 'grey' literature (e.g. technical reports, manufacturers' data sheets) was sought by a general internet search.

Searches were then broadened to literature on management options currently used for aquatic vegetation management in New Zealand (NZ) ponds, lakes and wetlands. In addition to mitigation measures, the review included practices aimed to avoid problems stemming from aquatic vegetation wherever possible (e.g. environmental manipulation, nutrient management, machinery hygiene).

2.2 Results

There were 14 references in the literature dealing with aquatic vegetation control in stormwater systems specifically. Most other references discovered during searches pertained to the establishment and utility of aquatic plants for the processing of stormwater run-off. A broader search was used to identify 35 management options employed in NZ aquatic systems, or those that may have relevance here. Not included were options that are unlikely to be available here now or in the near future (e.g. herbicides not registered for NZ use). Options and associated literature were organised into four main categories:

- **Biological Control**: the use of organisms to predate, infect or control the growth of the target pest, and/or to manipulate conditions leading to the control of the target pest.

- **Chemical control**: the use of pesticides and chemicals that either kill pests or inhibit their development, or the use of flocculants and nutrient inactivation products that are used to reduce nutrient loading, thereby decreasing algal growth.
- **Physical control**: removing pest biomass, or setting up barriers to their growth.
- **Preventative management**: avoiding invasion by pests, or actions to stop their populations developing.

3.0 Assessing control needs

In order to assess the benefits of control options for aquatic vegetation, a risk benefit analysis is recommended. Firstly, the problem should be defined before evaluating the need for control. What is the type of aquatic plant involved, where and when, and what problems are being caused to whom? As well as reactive options to problems, consideration should also be given to proactive actions to avoid the development of issues in the first place (Section 6.0), as these are usually the most cost effective. Terms shown in bold font in the text are defined in the glossary.

3.1 Defining algal problems

Algae are a natural component of aquatic environments that form the basis of aquatic food chains. For example, microalgae (i.e. phytoplankton) provide the food source for filter-feeders (e.g. mussels) and for **zooplankton**. An algae bloom, however, is rapid excessive growth of algae that can be visually conspicuous, potentially hazardous (if composed of a toxic **cyanobacteria** species), and/or be aesthetically unpleasant. Algae blooms that contain toxic cyanobacteria species are often referred to as **hazardous algae blooms (HABs)**. Furthermore, when an algae bloom collapses and large amounts of algae decompose, waters may become anoxic (greatly depleted in oxygen), leading to death of aquatic plants and animals.

Monitoring phytoplankton biomass (using a proxy such as **chlorophyll *a***) and species composition in a pond or lake is necessary to define the severity of an algae bloom and determine whether it is dominated by problematic (i.e. toxic) species. Sampling and analysis provide a measure of algal biomass as either **chlorophyll *a*** or **biovolume**, identify the presence of potentially toxic **cyanobacteria**, and can confirm of the presence of algal toxins. Although water sampling does not necessarily require specialist skills or equipment (although an appropriate method needs to be adopted), analysis does requires specialist instruments (usually lab based) and expertise. It is beyond the scope of this report to identify sampling protocols or recommend analysis methods and service providers, however, we refer readers to resources for lake monitoring (Burns et al 2000), and for monitoring cyanobacteria in recreational fresh waters (Wood et al 2009). It should be noted that wind may cause surface scums to accumulate in certain parts of the pond or lake so that there may be significant horizontal variability in phytoplankton concentration. Although stormwater management systems are unlikely to be used for recreational purposes, downstream effects should be considered. For example, blooms of toxic cyanobacteria in ponds that discharge into estuaries may result in high toxin levels in shellfish.

Highly eutrophic water bodies are particularly susceptible to algae blooms, and reduction of nutrient inputs from the catchment is a critical management action to reduce the incidence of algae blooms. However, nutrients may accumulate in sediments, and then be slowly released into the water column, enhancing algal growth even when nutrient inputs from the catchment have been minimised. In this situation, if algal growth has been determined to be a problem, then other management options may also be required to reduce the occurrence of algae blooms.

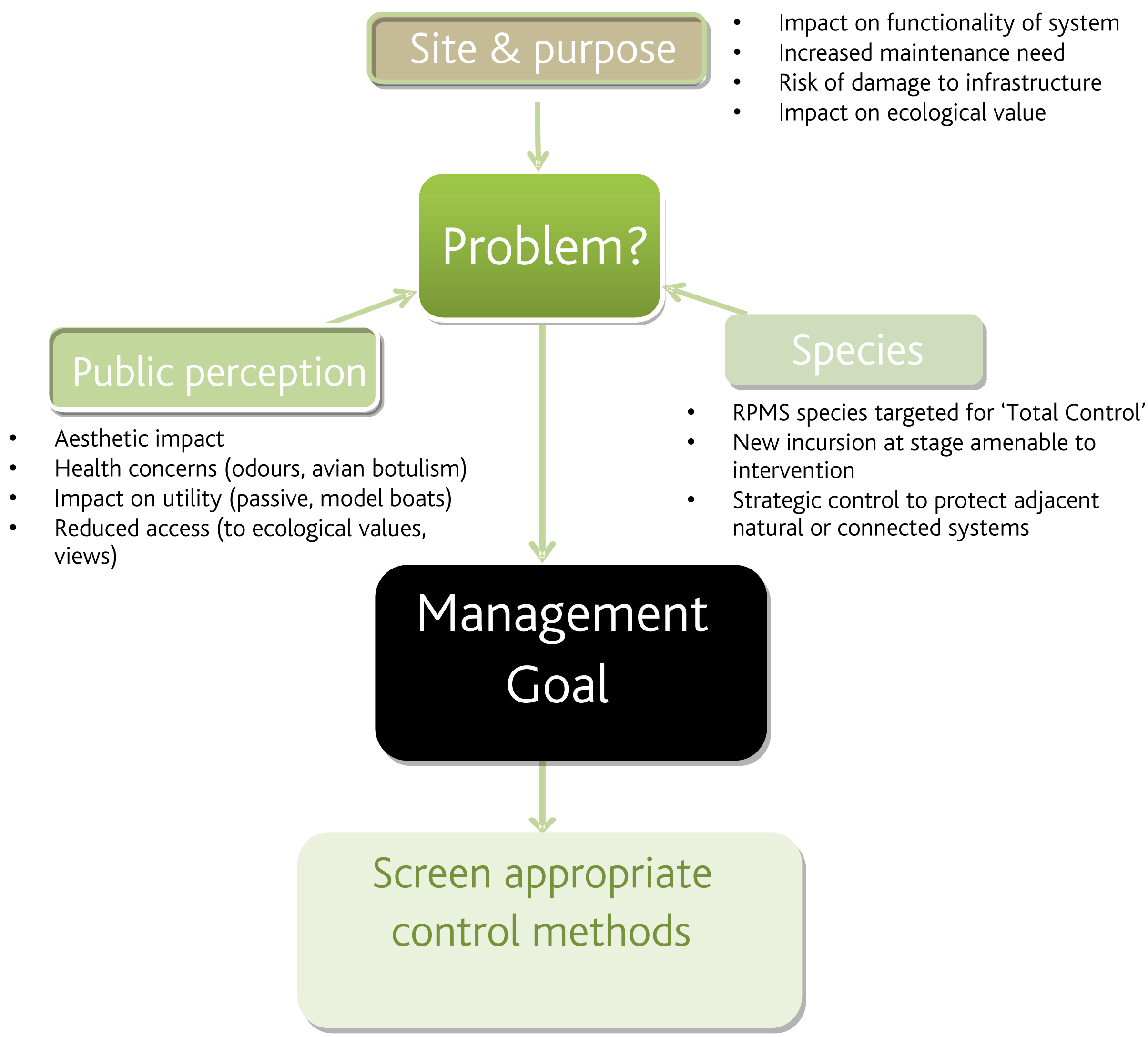
3.2 Defining plant problems

Elements to define the existence of a problem due to macroscopic aquatic plants are summarised in Figure 1. This evaluation of the problem (e.g. plant species, the size of infestation) is also required to identify the management goal (e.g. aquatic weed control and/or eradication) and for screening appropriate methods.

The site in question, its intended purpose and characteristics are important to define the existence of an aquatic plant problem. For example, stormwater systems in the Auckland region include constructed wetlands, where emergent and submerged plants are utilized for their ability to retard flows and filter and process stormwater runoff to improve water quality, for bank and bed stability, and to reduce water quantity through evapotranspiration. Native plants suited to the littoral shelf and pond areas are encouraged (Auckland Regional Council 2003), but some level of alien or weedy species may be acceptable, especially if similar functions are provided. However, the drainage and processing of stormwater flows may be compromised by undesirable growths, such as floating sudds created by yellow flag (*Iris pseudacorus*), or dense, ramified beds of manchurian wild rice (*Zizania latifolia*). Other impacts on the functionality of wetland and pond systems might include submerged weedbeds that obstruct risers or outlets or that block debris screens, and marginal vegetation that occupies spill ways, or that damages embankments.

More difficult to subjectively assess are public perceptions, and impacts on aesthetic values of stormwater systems. These may include a preference for open water areas unoccupied by surface-reaching submerged plants, concerns that water 'stagnates' or trash accumulates in weed beds, opposition to the loss of water views and the importance of 'access' to nature. Public concerns for human or ecological health may be fuelled by odours from decaying plant material, or outbreaks of avian botulism in waterfowl. Although the type of botulism toxin that causes waterfowl die-offs is of low risk for human and pet health (Friend and Franson1999), the sight of dead and dying birds can be distressing. The link between botulism outbreaks and the need for weed management is not direct, and rather the most effective management steps to reduce outbreaks involves the disposal of any waterfowl carcass before other birds can feed on maggots which re-cycle and concentrate the toxin (Friend and Franson 1999). Although boating, swimming and fishing are actively discouraged on the stormwater ponds and wetlands of Auckland region, larger open ponds with good access may be utilized for model boating. Surface-reaching weed beds will be in conflict with this activity.

Figure 1 Main elements to defining a problem relating to aquatic plant management in stormwater systems.



Most frequently, issues with the aquatic plants are driven by invasive or pest species. The Regional Pest Management Strategy (RPMS) for the Auckland region (Auckland Regional Council 2007) lists 14 aquatic plant pests that are managed for 'Total Control' (Table 1). These species are of 'limited distribution or density within the Auckland region, or defined areas of the region', and are 'considered to be of high potential threat to the region'. AC assumes responsibility for funding and implementing appropriate management programmes for total control species.

A further 27 aquatic plant pests are listed as 'Surveillance' plants (Table 1), that are recognised to threaten the environment. AC encourages landowners and occupiers to remove these surveillance species from their properties, although there is no legal requirement to enforce this. AC also enforces a ban on sale, propagation, distribution and exhibition of these pests.

The discovery of a new pest plant incursion might present an opportunity to head off future issues with an early intervention. Control of a recognised pest species may also be higher priority where the control reduces risk to other connected or adjacent systems, especially natural waterways. Stormwater ponds are noted for their potential to be a source of weed spread in the Auckland region (Young and Carter 2005).

Consideration of all these factors, together with appropriate weighting for the intended purpose of the stormwater system, should be used to determine if a problem exists and to identify an appropriate management goal. Goals could include the eradication of a pest species, or mitigation of the level of impact from a plant problem, or no action may be deemed necessary.

Table 1 Aquatic plant species listed on the Regional Pest Management Strategy (RPMS) for the Auckland region.

Species	Common name	RPMS status
**Egeria densa*	egeria	Total Control
Gymnocoronis spilanthoides	senegal tea	Total Control
Hydrocleys nymphoides	hydrocleys	Total Control
Lythrum salicaria	purple loosestrife	Total Control
Nymphoides geminata	mashwort	Total Control
Osmunda regalis	royal fern	Total Control
Sagittaria montevidensis, *S. platyphylla* & *S. sagittifolia*	arrowhead, sagittaria	Total Control
†*Spartina alterniflora*, *S. anglica* & *S. x townsendii*	spartina	Total Control
Typha latifolia	great reedmace	Total Control
Zizania latifolia	manchurian wild rice	Total Control
All *Equisetum* spp.	horsetail	Surveillance
Alocasia macrorrhiza syn. *A. brisbanensis*	elephant's ear	Surveillance
Alternanthera philoxeroides	alligator weed	Surveillance
Arundo donax	giant reed	Surveillance
Ceratophyllum demersum	hornwort	Surveillance
Eichhornia crassipes	water hyacinth	Surveillance
Glyceria maxima	reed sweet grass	Surveillance
Gunnera tinctoria	chilean rhubarb	Surveillance
Hydrilla verticillata	hydrilla	Surveillance
Iris pseudacorus	yellow flag	Surveillance
Lagarosiphon major	lagarosiphon	Surveillance
Ludwigia peploides subsp. montevidensis	water primrose	Surveillance
Lycopus europaeus	gypsywort	Surveillance
Marsilea mutica	nardoo	Surveillance
Menyanthes trifoliata	bog bean	Surveillance
Myriophyllum aquaticum	parrots feather	Surveillance
Nuphar lutea	water lettuce	Surveillance
Nymphaea mexicana	mexican water lily	Surveillance
Nymphoides peltata	fringed water lily	Surveillance
Phragmites australis	phragmites	Surveillance
Pistia stratiotes	water lettuce	Surveillance
Potamogeton perfoliatus	clasped pondweed	Surveillance
Salix cinerea	grey willow	Surveillance
Salvinia molesta	salvinia	Surveillance

Utricularia gibba	bladderwort	Surveillance
Vallisneria australis (formerly *V. gigantea* & *V. spiralis*)	eel grass	Surveillance
Zantedeschia aethiopica	arum lily, green goddess	Surveillance

*Total control on Great Barrier Island, elsewhere surveillance.

†Total control in Waitemata and Manukau Harbours, and all waterbodies of the east coast of the Auckland region, elsewhere surveillance.

4.0 Decision making framework

A flow-diagram (Figure 2) outlines the main steps for screening control options for an aquatic vegetation problem.

This should firstly consider what the target vegetation is (e.g. algae or higher plants, emergent or submerged plants), while it may also be necessary to identify the species of problem plant as some options are specific to some weeds and not others (See Section 3.0).

The next step is to consider the type of system the problem is occurring in, its purpose and major characteristics, such as size and configuration. To assist in this step, a categorisation of stormwater systems is provided (Figure 3) that reflects some operational and environmental constraints on the choice of control option. This considers the type of system; wet pond, wetland or receiving streams (those heavily influenced by stormwater volumes). Features that determine the type of stormwater device employed include slope, catchment area and land use, soil permeability and stability, space, and depth of the water table (Auckland Regional Council 2003). Recommended pond and wetland designs (Auckland Regional Council 2003) also differ in terms of maximum depth, depth variation and therefore the degree of flows/flushing.

The primary purpose of stormwater systems should also be considered. Wet ponds for water quantity management divert, retain and slowly release volumes of stormwater, while other ponds are for water quality improvement primarily via physical processing, i.e. particle settlement. Submerged vegetation is not required for water quality processing in ponds, but may provide a beneficial role in some situations. Constructed wetlands do require vegetation presence for the purpose of water quality improvement. Most ponds and wetlands are 'offline' i.e. do not receive flows from, or drain to, natural streams, or only drain to ephemeral watercourses. The main purpose of any receiving streams is to convey quantities of stormwater, but a secondary purpose would be retention of ecological values. Amenity value is a recognised purpose for the stormwater systems in the Auckland region, and control options will need to be considered for their public acceptability.

The size or configuration of systems may determine their amenity to certain options (e.g. shading from tree planting). Method of construction of systems may be relevant, for instance, excavated ponds may have ground water inflows that dilute herbicides, or are less amenable to drainage and drying. The location of the system might determine options in the case of coastal ponds that are amenable to saline intrusions or tidal flushing (see 5.28, Periodic saline intrusions).

Figure 2 Identified steps to screen appropriate control options for aquatic vegetation management.

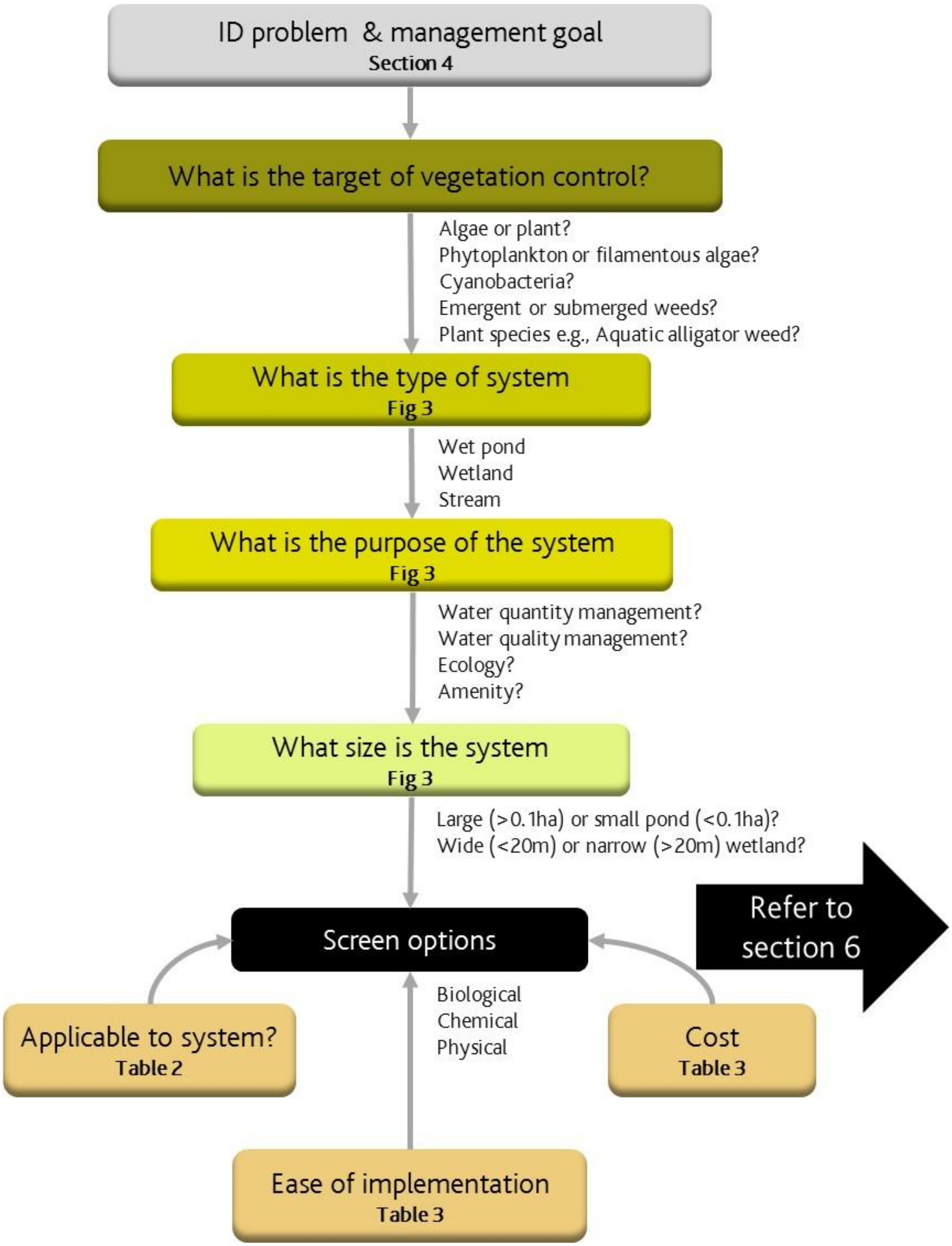
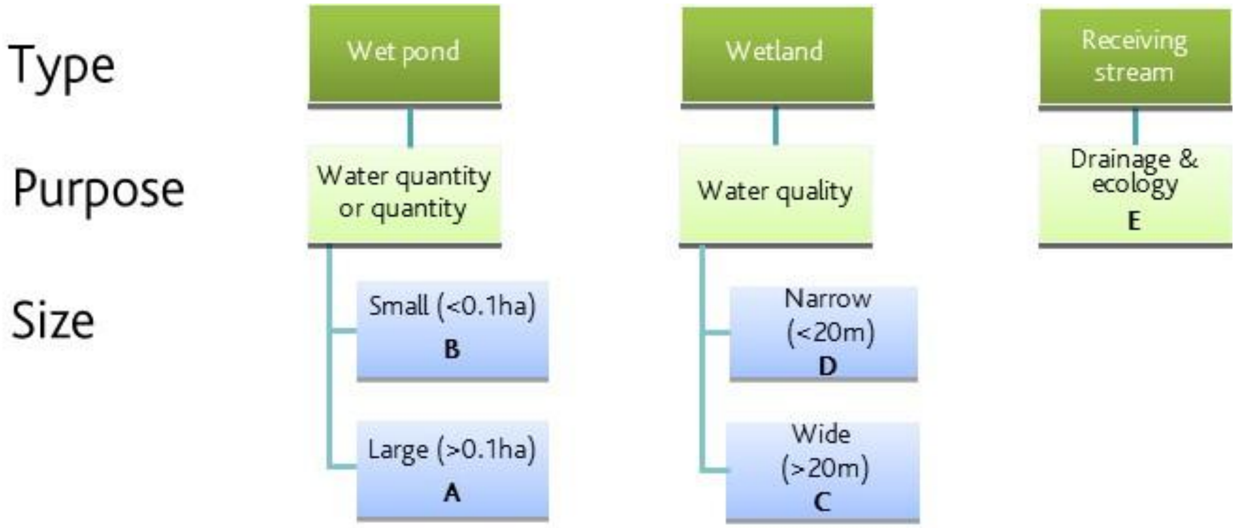


Figure 3 Categorisation of five types of stormwater systems (A to E) relevant to identifying appropriate aquatic plant management options.



Control options can be screened using the reference tables provided below. These reference tables should be used to shortlist possible options, with the final choice of option made in reference to option descriptions in Section 5.0, which provide sufficient information for individual site and species-specific considerations. In the following tables options are grouped under categories of biological, chemical or physical, with options following the same order in Section 5.0.

The target vegetation type for each option, and whether the option is applicable for the stormwater system in question is indicated in Table 2. Five categories of stormwater system (Figure 3) are identified, and a tick (✓) in a column indicates it is likely to be an applicable option for that system, while a cross (×) suggests it is not applicable, and a question mark (?) indicates uncertainty or the need for site or species-specific consideration. An asterisk (*) identifies those options that can be applied to parts of a system, as opposed to treatments that are likely to influence the entire aquatic system.

Other considerations for the screening of options include an indication of costs per ha, over the lifetime of the intervention (maximum of 25 years). These costs (Table 3) are indicated as Low (< $10k), Moderate ($10k to $25k), High ($25k to $50k) or Very high (> $50k). Caveats to these indicative costs are indicated in Table 3. Please refer to Section 5.0 for more information.

Lastly, an indication of the ease of implementation of the option is provided. This is not related to the likely effectiveness of an option which can vary on site and species-specific basis. Please refer to Section 5.0 for more information.

Table 2 Identification of control options and suitability for target vegetation, together with applicability to each category of system. ∗ Options that can be applied to parts of a system rather than entire aquatic system.

Target		Option	Large wet pond	Small wet pond	Wide wetland	Narrow wetland	Receiving stream
			A	B	C	D	E
Aquatic alligator weed	Biological	Classical biological control	✓	✓	✓	✓	✓
Submerged weeds		*Mycoherbicide	?	?	?	?	?
Submerged weeds/filamentous algae		Grass carp	✓	✓	×	×	×
Cyanobacteria		Silver carp	?	?	×	×	×
Phytoplankton		Microbial products	?	?	×	×	×
Phytoplankton		Barley straw	?	✓	×	×	×
Phytoplankton		Macrophyte restoration	?	✓	×	×	×
Phytoplankton		Pest fish removal	✓	✓	?	?	×
Phytoplankton		Zooplankton or invertebrate grazers	✓	✓	?	?	×
Phytoplankton/filamentous algae		Waterfowl management	✓	✓	?	?	?

Target		Option	Large wet pond	Small wet pond	Wide wetland	Narrow wetland	Receiving stream
			A	B	C	D	E
Phytoplankton/filamentous algae /submerged weeds	Chemical	Chelated copper	?	?	×	×	×
Emergent plants		* Glyphosate isopropylamine	✓	✓	✓	✓	✓
Submerged weeds		Diquat	✓	✓	×	×	×
Submerged weeds		Endothall	✓	✓	×	×	×
Emergent weeds		*Restricted herbicides	?	?	?	?	?
Terrestrial weeds		*Natural herbicides	?	?	?	?	?
Phytoplankton		Nutrient inactivation products	✓	?	×	×	?
Phytoplankton		Flocculation	✓	✓	?	?	?
Phytoplankton/filamentous algae /submerged weeds	Physical	*Physical Shading	?	✓	?	?	✓
Phytoplankton/filamentous algae /submerged weeds		Shading by dyes	?	✓	×	×	×
Submerged weeds/filamentous algae		*Manual harvesting	✓	✓	✓	✓	×
Submerged weeds/filamentous algae		Mechanical harvesting	✓	?	×	×	×

Target		Option	Large wet pond	Small wet pond	Wide wetland	Narrow wetland	Receiving stream
			A	B	C	D	E
Emergent/submerged weeds	Physical	*Mechanical excavation	✓	✓	✓	✓	✓
Emergent weeds		*Mowing	✓	✓	✓	✓	✓
Submerged weeds		*Bottom lining	✓	✓	✓	✓	×
Submerged weeds		*Suction dredging	✓	✓	✓	✓	✓
Submerged weeds		Water level drawdown	✓	✓	×	×	×
Phytoplankton/submerged weeds		Periodic saline intrusions	✓	✓	×	×	×
Phytoplankton (submerged weeds?)		*Substrate capping	?	✓	?	?	?
Phytoplankton (submerged weeds?)		*Sediment removal	✓	✓	?	?	?
Phytoplankton		Aeration and artificial destratification	✓	?	×	×	×
Phytoplankton		UV lights	?	?	?	?	?
Phytoplankton		Ultrasonication	?	✓	×	×	?
Phytoplankton		Wave attenuation barriers	✓	?	×	×	×
Phytoplankton		Hydraulic flushing	?	?	?	?	?

Table 3 Identification of control options and indicative costs (Low =< $10k, Moderate = $10k to $25k, High =$25k to $50k, Very high => $50k), together with an indication of the ease of implementation. [a] Costs reduced if community group involvement is possible. † Costs do not include approvals, consents, compliance monitoring, or reporting. ? Insufficient information to guide estimate.

Target		Option	Cost	Ease of implementation
Aquatic alligator weed	Biological	Classical biological control	Low	Easy
Submerged weeds	Biological	Mycoherbicide	†Moderate	Moderate to difficult
Submerged weeds/filamentous algae	Biological	Grass carp	†High	Moderate
Cyanobacteria	Biological	Silver carp	†High	Moderate
Phytoplankton	Biological	Microbial products	Moderate to high	Easy
Phytoplankton	Biological	Barley straw	Low	Easy
Phytoplankton	Biological	Macrophyte restoration	Moderate to high	Moderate to difficult
Phytoplankton	Biological	Pest fish removal	High	Moderate
Phytoplankton	Biological	Zooplankton or invertebrate grazers	?	Difficult
Phytoplankton/filamentous algae	Biological	Waterfowl management	Low	Easy

Target		Option	Cost	Ease of implementation
Phytoplankton/filamentous algae /submerged weeds	Chemical	Chelated copper	Moderate	Moderate
Emergent plants		Glyphosate isopropylamine	Low	Easy
Submerged weeds		Diquat	†Moderate	Moderate
Submerged weeds		Endothall	†Moderate	Moderate
Emergent weeds		Restricted herbicides	†Moderate	Moderate
Terrestrial weeds		Natural herbicides	Very high	Easy
Phytoplankton		Nutrient inactivation products	High	Difficult
Phytoplankton		Flocculation	High	Difficult

Target		Option	Cost	Ease of implementation
Phytoplankton/filamentous algae /submerged weeds	Physical	Physical Shading	Moderate to high	Easy to moderate
Phytoplankton/filamentous algae /submerged weeds		Shading by dyes	Very high	Easy
Submerged weeds/filamentous algae		Manual harvesting	[a]Moderate	Easy
Submerged weeds/filamentous algae		Mechanical harvesting	High	Easy to moderate
Emergent/submerged weeds		Mechanical excavation	Low	Easy
Emergent weeds		Mowing	Moderate	Easy
Submerged weeds		Bottom lining	High	Moderate to difficult
Submerged weeds		Suction dredging	Moderate	Moderate to difficult
Submerged weeds		Water level drawdown	Low	Easy to moderate
Phytoplankton/submerged weeds		Periodic saline intrusions	Low	Easy to moderate
Phytoplankton (submerged weeds)		Substrate capping	Moderate	Moderate
Phytoplankton (submerged weeds)		Sediment removal	High to very High	Moderate

Target		Option	Cost	Ease of implementation
Phytoplankton		Aeration and artificial destratification	High to very High	Moderate to difficult
Phytoplankton		UV lights	High to very High	Moderate to difficult
Phytoplankton		Ultrasonication	High	Moderate to difficult
Phytoplankton		Wave attenuation barriers	?	Moderate
Phytoplankton		Hydraulic flushing	High	Moderate to difficult

5.0 Management options

This section summarises relevant information on the management options, listed under biological control, chemical control, and physical control. The information presented for each option is based on publications, reports and other literature (Section 2.0; Literature review), as well as expert knowledge and authors' familiarity with emerging technologies or practices that are not yet available in the literature. Overseas information has been reviewed for relevancy to the NZ situation (e.g. available registered herbicides and chemical nutrient management options).

The purpose of this section is to provide sufficient information to guide the specific selection of management option(s) over a range of situations. Information includes a brief description of what the option entails, the level of information available, and the likely duration of control. The applicability of the option to stormwater systems and any constraints to use are considered, as is the potential for incorporating other options in an integrated management approach. The extent of use of the control option and outcomes are briefly reviewed for NZ and overseas. Implementation of the option and any on-going effort are described at a generic level, as are any related practical considerations. Finally indicative costs are considered across various stages of the life-cycle of the management option (as annual or one-off costs), with an overall assessment of costs over the lifetime of the intervention, up to 25 years. Costs are indicated as Low (< \$10k), Moderate (\$10k to \$25k), High (\$25k to \$50k) or Very high (> \$50k). Terms shown in bold font in the text are defined in the glossary.

Biological control

Biological control options for plants

5.1 Classical biological control

5.1.1 Description and overview

Classical biocontrol of aquatic pest plants in NZ is currently limited to the use of **arthropods** as control agents on alligator weed. A flea beetle, *Agasicles hygrophila*, and a moth, *Arcola* (formerly *Vogtia*) *malloi*, have been widely released in the upper North Island (Stewart et al 1999; Winks 2007a and b).

Flea beetle adults and larvae feed on the leaves and stems of aquatic alligator weed (Winks 2007 a). Caterpillars of the moth graze on and destroy stems of aquatic alligator weed, while the adults are general nectar feeders (Winks 2007b).

5.1.2 Application - *In what situations can the option be applied?*

These agents are best applied to large, aquatic beds of alligator weed, where they may contribute to some level of suppression (van Oosterhout 2007).

5.1.3 Constraints - *In what situations can the option not be applied?*

This classical biological control will not achieve eradication of alligator weed, and is limited in geographic application. The present strain of flea beetle is limited by temperatures in most parts of NZ and therefore is not suitable for widespread control of alligator weed (Stewart et al 1999), while the effectiveness of the moth is also somewhat restricted by temperature (Hayes 2007).

Neither biocontrol agent will establish on or control alligator weed growing on land (as opposed to weed growing over water) (Winks 2007a and b). Flowing waters that periodically flood over the alligator weed beds are also not suitable for flea beetle or the moth larvae due to the removal and loss of these agents (Winks 2007a and b).

There is evidence in Australia that flea beetle is less successful in drains and small, ephemeral waterways than on larger, permanent water bodies (Julien and Bourne 1988).

5.1.4 Requirements - *What other options / practices might be required in conjunction with this option?*

Other control methods (e.g. herbicides) are not recommended at sites where biocontrol agents have been released (Winks 2007a and b).

5.1.5 Track record - *Where has this option been successful / unsuccessful?*

These two agents have been used in the USA and Australia, and the flea beetle in China and Thailand (Hayes 2007). In NZ, some control of alligator weed mats has been achieved around large lakes and ponds (Hayes 2007). But the agents are of little value in eradication programmes and have done little if anything to slow the spread of this plant.

5.1.6 Implementation - *Methods employed*

Flea beetle will naturally disperse from adjacent infestations if these are close, and the flea beetle is widely dispersed throughout the range of alligator weed in Auckland. If not already present at the site, they need to be collected from a source site (c. 100 adults) and actively released at new infestations (Winks 2007a). Likewise, adult moths can move short distances, otherwise stems occupied by caterpillars can be collected in late summer (kept damp and cool) and introduced to the site (Winks 2007b). Introductions should be to high density infestations of alligator weed only.

5.1.7 Operation, maintenance, monitoring and reporting

Checks after initial release will determine if establishment has been successful.

5.1.8 Practical considerations - *e.g. social issues, access constraints, consent requirements, health and safety concerns*

This option is less effective than chemical control options, but may be more acceptable to the public under some circumstances. There are no consent requirements or health and safety concerns.

5.1.9 Financial costs

Cost estimate (per ha treated unless stated otherwise)

(i) Start-up / implementation (once-off): Low

(ii) Operation & maintenance (annual): Low

(iii) Monitoring (annual): Low

(iv) Decommissioning, if relevant (once-off): Low

(v) Overall cost over the lifetime of the intervention (maximum of 25 years): Low

5.2 Mycoherbicide

5.2.1 Description and overview

The fungus Mt (*Mycoleptodiscus terrestris*) is a naturally occurring plant pathogen in aquatic and waterlogged systems (Shearer 1997). Mt has been investigated in the USA and NZ for its potential as an inundative biocontrol agent for submerged aquatic plants. Unlike classical biological control where a self-sustaining balance is formed in the environment between the agent and the pest plant/host, the inundative approach relies on greater numbers (or amount) of the agent being introduced, which overwhelms the pest/host (Shearer 2008).

In NZ Mt has been isolated off both an invasive aquatic plant species (hornwort) and desirable native plants (Hofstra et al 2009 and 2012), indicating that a native host association exists in NZ lakes. Material is cultured (Figure 4), and applied to target submerged weed beds. The fungi infects plant cells and causes them to break down, after which the Mt levels also decline. Research is currently underway to test **virulence** and **efficacy** in the field.

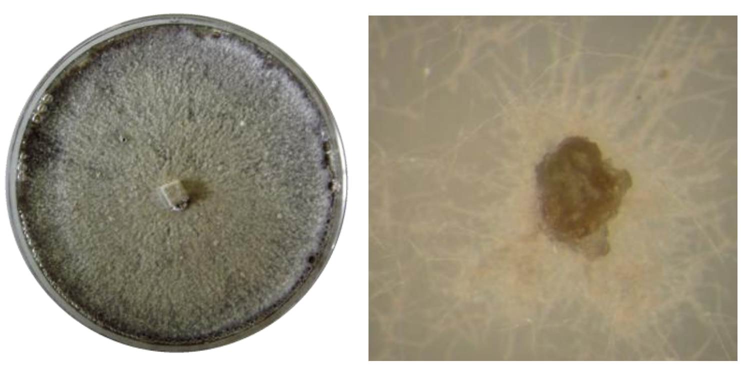


Figure 4 *Mycoleptodiscus terrestris* in laboratory culture (Photo: Deborah Hofstra, NIWA).

5.2.2 Application - *In what situations can the option be applied?*

Mt would be applicable to dense weed beds in still water.

5.2.3 Constraints - *In what situations can the option not be applied?*

Mt is unlikely to be suitable for use in flowing water.

5.2.4 Requirements - *What other options / practices might be required in conjunction with this option?*

It is likely that Mt could be used in conjunction with known aquatic herbicides (both at lower concentrations) for improved efficacy and to lower the environmental herbicide load (Netherland and Shearer 1996; Nelson et al 1998).

5.2.5 Track record - *Where has this option been successful / unsuccessful?*

In laboratory and **mesocosm** scale studies in the USA and NZ Mt (different **isolates** and formations) has successfully controlled a range of target aquatic weed species. Field efficacy has not been as successful to date (Shearer 1998).

5.2.6 Implementation - *Methods employed*

Not yet commercially available. However, similar methods to those used for an aquatic herbicide are employed, e.g. liquid spray or broadcast application of solid product.

5.2.7 Operation, maintenance, monitoring and reporting

Not yet commercially available.

Operation, maintenance, monitoring and reporting are likely to be similar to the use of an aquatic herbicide. For example, the monitoring requirements would depend on the level of weed control required and the time taken for plants to regrow back to nuisance levels following an application.

5.2.8 Practical considerations *- e.g. social issues, access constraints, consent requirements, health and safety concerns*

Biological control is generally perceived more favourably by the public than the use of herbicide. However further research is still required on how to achieve efficacy in field trials. The use of a mycoherbicide will still require a resource consent as it is a living organism intended for application to or in water, as for herbicide.

5.2.9 Financial costs

Not yet commercially available, but expect that costs would be similar to those for herbicide use.

Cost estimate per ha*

(i) Start-up / implementation (once-off): Moderate.
(ii) Operation & maintenance (annual): Low
(iii) Monitoring (annual): Low
(iv) Decommissioning, if relevant (once-off): NA
(v) Overall cost over the lifetime of the intervention (maximum of 25 years): Moderate
* Costs do not include approvals, consent costs, public notifications, compliance monitoring, or reporting, which are likely to vary on a situational basis.

5.3 Grass carp

5.3.1 Description and overview

Grass carp (*Ctenopharyngodon idella*, white amur) are introduced **herbivorous** fish (Figure 5) that are bred in NZ for submerged vegetation control (Rowe and Schipper 1985; Rowe and Hill 1989). They are unlikely to **naturalise** (Rowe and Schipper 1985; Baker and Smith 2006).

Grass carp are non-selective grazers (Dibble and Kovalenko 2009). Provided they are stocked at a sufficient density, grass carp will eradicate some problem weeds and provide **total control** of submerged vegetation for the duration of time the fish are present in a system.

Total control can be evident within two years of stocking. If conditions favour the long-term survival of the fish, then their effects will last for 20 years or more until the fish die out or are otherwise removed. Even when all aquatic vegetation is removed, a reduced number of fish can survive by consuming fallen leaves, marginal grasses and epiphytic algae (Authors' observations).

Figure 5 Grass carp captured after eradication of weeds in a Hawkes Bay lake (Photo: R. Wells, NIWA).

5.3.2 Application - *In what situations can the option be applied?*

Use of grass carp may be suitable where eradication of submerged weeds is sought, or where vegetation biomass needs to be kept at an on-going low level. Grass carp can eradicate weeds that reproduce only by vegetative means (e.g. egeria, hornwort), if there are no upstream sources that can re-infest the system.

Vegetation control should be achieved after two summers under the warm water temperatures in Auckland region. Active feeding occurs above >15–17°C and grass carp are known to feed intensively at 20–23°C, consuming 100% of their body weight per day (Rowe and Schipper 1985).

Grass carp need to be kept within the treatment area, therefore if there are inlets, outlets or flood spillways these will need to be secured with screens, usually constructed from metal bars (Hofstra 2011) and designed for the size of fish. Screens will need to cope with flood flows and there may be a requirement to provide passage for native fish such as eels, which can add costs to screen design and construction.

5.3.3 Constraints - *In what situations can the option not be applied?*

Grass carp will not be a suitable option where systems are managed for beneficial influences of wetland or submerged vegetation. Partial or selective vegetation control is unlikely (Cassani 1996; Bonar et al 2002; Hofstra 2011), due to the dynamic nature of vegetation growth and shifting equilibriums with grass carp grazing. For instance, most emergent wetland plants are consumed if water depth is sufficient for fish to access marginal areas (c. ≥0.5 m) and these plants are likely to be removed unless fenced exclosures are provided to protect these areas (Rowe et al 1999).

Plants that are not grazed include short-growing turf plants (e.g. *Glossostigma* and *Lilaeopsis* species) and plants floating on the water surface (e.g. *Lemna* or *Azolla* species) (Rowe and Schipper 1985).

The time required to obtain approvals for stocking means that grass carp are unlikely to provide a rapid response solution to new weed incursions or issues, unless approvals are pre-arranged.

Grass carp are not suitable for use in systems from which they are likely to escape. For example, using screens in flowing-water situations would be particularly challenging due to debris build-up, need for regular screen maintenance, and possible breaches or damage.

If fish escape, re-stocking may be costly. Nevertheless, the risk of escaped grass carp having unwanted impacts elsewhere in the catchment is low for stormwater systems in Auckland region, where most are within small catchments with short drainage distances to the sea. Grass carp are not a sea-going fish (Cross 1970), they do not enter salt water voluntarily and can tolerate only a few hours at 50% sea water. Spread to other catchments via marine pathways is not documented and considered unlikely to occur.

Grass carp are unsuitable for stormwater systems where water quality prevents long-term grass carp survival. Low oxygen and/or acid pH (3-4) has been thought responsible for grass carp kills in agricultural drains within Waikato Region (Wells et al 2003; D Rowe NIWA pers. comm.). A lethal dissolved oxygen threshold of ≤0.5 mg L^{-1} (5.5% at 20˚C) is reported for grass carp (Sutton, 1985), however, grass carp are able to cope with temporary low oxygen by gulping air at the surface (Hofstra and Clayton 2012).The presence of (acid) peat soils in the catchment of systems may also be considered a risk factor.

Effects of grass carp grazing of vegetation on water quality are not clear, with the literature showing contradicting findings reported from experimental, monitoring or modelling work. Dibble and Kovalenko (2009) concluded that current data are not sufficient to adequately answer important questions about use of grass carp.

Grass carp use will need to be carefully assessed for systems with high ecological values. They can indirectly alter the feeding habitats and spawning behaviour of some fish species which depend on using the habitats provided by weed beds. Mitchell (1986) and Rowe (1984) showed changes to other species of fish with macrophyte removal leading to increased predation (by shags in particular), a lower number of smelt and a dietary shift in both bullies and smelt from zooplankton to chironomids. The removal of weed by grass carp increased the abundance of both bullies and dwarf inanga in a Northland lake (Rowe et al1999), reflecting an increase in the benthic food chain. Grass carp may reduce local waterfowl (especially swan populations) by removing food sources. Many invertebrate species also require aquatic vegetation during part of their life cycle and their diversity and abundance may be reduced when grass carp are used.

5.3.4 Requirements - *What other options / practices might be required in conjunction with this option?*

An initial herbicide treatment (see 5.13; Diquat or 5.14; Endothall) can be made prior to grass carp stocking to ensure faster control, a reduced number of grass carp and lower costs.

Currently the re-capture or *in situ* euthanasia of grass carp when vegetation control aims are achieved is challenging, especially in large, or deep lake and pond systems, although retrieval in drains is more feasible. Even so, fish removal may be costly. Removal options include netting, piscicides in bait form or whole-of-waterbody application, or drainage, or combinations of these. A whole-of-waterbody dose of the piscicide rotenone (see 5.8; Pest fish removal) has been used to completely remove grass carp from a NZ lake (Rowe and Champion 1994). Rotenone affects all fish species, although there is potential to revive and restock desired species such as lacustrine common bullies and smelt. **Diadromous** fish species will re-establish naturally. The use of baits involves training grass carp to feed on floating, food pellets, which are then substituted for Prentox pellets containing a piscicide (rotenone or antimycin) (Rowe 1999). This method has had some success overseas and on one occasion in NZ, but grass carp rapidly learn to avoid such baits meaning a repeat treatment is ineffective. Other species of fish (e.g. eels, rudd and perch) also outcompete grass carp to consume floating baits so it is not applicable in waters containing these species (D. Rowe NIWA pers. comm.). **Trammel nets** made from fine monofilament are most successful physical capture method.

5.3.5 Track record - *Where has this option been successful / unsuccessful?*

Successful aquatic weed control operations using grass carp in small NZ ponds and lakes (Rowe and Champion 1994, Clayton et al 1995) include Waihi Beach Reservoir, Bay of Plenty (1.92 ha), Parkinson's Lake, South Auckland (2 ha), Elands Lake, Hawke's Bay (4 ha), Lake Waingata, Pouto (12 ha), Wainamu, Auckland, and more recently Lakes Tutira and Opouahi, Hawkes Bay.

In NZ's agricultural drains, results are mixed. Grass carp successfully maintained weed control for three years in the Mangawhero Drain (D. Rowe NIWA pers. comm.) and were successful in a Waikato Drain (Wells et al 2003). But long-term success of control in drain environments is yet to be demonstrated because most grass carp in these systems suffer frequent fish kills from low oxygen, acid pH, and predation, or escaped during floods.

In the Auckland region, newspaper articles[1] suggest grass carp stocking into stormwater ponds started in 2005 at Hayman Park, Manukau City and Wattle Farm Reserve, Wattle Downs. The control of weed there resulted in more grass carp being released into Pakuranga, Botany and

[1] http://www.times.co.nz/news/carping-to-beat-the-weeds.html

http://www.stuff.co.nz/auckland/local-news/eastern-courier/3138806/Unhappy-end-for-council-s-carp

Flat Bush ponds, to approximately 15 Auckland sites. However, fish kills have also been reported that suggest conditions in ponds may be challenging for fish survival at times.

5.3.6 Implementation - *Methods employed*

Usually, fish over 250 mm **fork length** (FL) are stocked as these are less prone to predation by shags. Required fish stocking densities are >50 fish per ha of weed covered lake area to ensure plant consumption exceeds the growth rate of the vegetation. Stocking more fish will achieve faster vegetation control (Hofstra and Clayton 2012) and allow for some mortality. Once vegetation biomass is reduced, a lower density of fish can still exert total control.

Stocking planned for spring would maximise grazing over the subsequent summer period.

Grass carp use is subject to approvals from Department of Conservation (DOC) and Ministry for Primary Industry (MPI). An assessment of the risks is required to be supplied to the Minister of Conservation (through DOC) in the form of an **AEE** (Assessment of Environmental Effects) report. Approvals may also be required from regional Fish and Game Councils, and consultation with relevant iwi and other affected parties (landowners, recreational groups etc.,) is required for the AEE. An **Operational Plan** is also required to determine containment, monitoring and long-term fish management requirements. Guidelines on the application process and application forms are available from DOC permissions staff.

5.3.7 Operation, maintenance, monitoring and reporting

On-going effort after initial stocking is relatively low, unless re-stocking is required. Clearance of screens should be done at intervals of 2 to 3 months or immediately following large rainfall events. Annual assessments of minimum grass carp numbers can be based on late-summer observations of fish present at the water surface (Hofstra and Rowe 2008). Alternatively, the continued removal of vegetation would indicate grass carp survival at sufficient numbers.

Monitoring of key water quality parameters at critical times (e.g. mid-summer dissolved oxygen and temperature) is advisable to explain any observed fish kills in stormwater systems.

An additional, one-off effort would be required to remove grass carp.

5.3.8 Practical considerations - *e.g. social issues, access constraints, consent requirements, health and safety concerns*

Public concern over grass carp use in artificial systems is likely to be lower than for natural waterbodies, and public education about risks may be helpful. It will be important to clearly identify the outcome sought (e.g. eradication of weeds, eventual restoration of native vegetation, or on-going vegetation control), so that public expectations are managed.

5.3.9 Financial costs

Annual leasing fees per site or per ha are offered at $2,250 per annum for contracts of 5 years duration, and $1,500 per annum for contracts of 10 years duration, which include manipulations of fish number, surveys and reports (B. Jamieson NZ Waterways Restoration Ltd 2013 pers. comm.). Indirect costs may be site-specific, and include consultation and approvals that can be time consuming (Hofstra and Clayton 2012) and expensive. Additional costs are for containment screens and signage to prohibit fishing.

Costs below are on a direct, per ha basis, assuming a maximum of a ten year time for eradication of a problem weed.

Cost estimate per ha*

(i) Start-up / implementation (once-off): Low

(ii) Operation & maintenance (annual): Low

(iii) Monitoring (annual): Low

(iv) Decommissioning, if relevant (once-off): Low

(v) Overall cost over the lifetime of the intervention: High (25 year period)

* Costs do not include consultation, or approvals, which are likely to vary on a situational basis.

Biological control options for algae

5.4 Silver carp

5.4.1 Description and overview

Silver carp (*Hypophthalmichthys molitrix*) are an introduced **planktivorous** fish (Figure 6) that have been artificially bred in NZ specifically for their potential to control phytoplankton (Rowe 2010). As with grass carp, they do not breed in small lakes, ponds or streams, and fish must be stocked into systems at unnaturally high densities to achieve control.

Silver carp feed on suspended particles greater than 0.01 mm in size using specialised filtering apparatus on their gills. Considered opportunistic feeders, silver carp can consume a range of **phytoplankton** as well as zooplankton and **detritus**. Silver carp will consume **cyanobacteria**, including problematic taxa such as *Anabaena* and *Microcystis* (Xie and Liu 2001; Ke et al 2009; Ma et al 2012).

Most information presented here comes from a literature review carried out by Rowe (2010), who concludes the greatest potential for silver carp use in NZ is in cyanobacteria control within eutrophic lakes and ponds. This is the focus of the following information.

Figure 6 Silver Carp (Photo: Dr R M McDowall, NIWA).

5.4.2 Application - *In what situations can the option be applied?*

Silver carp may have some application for cyanobacteria control in stormwater systems that experience regular blooms of these nuisance algae. However, Rowe (2010) noted that there is very little information on the use of silver carp in temperate environments compared to warm tropical areas. The information available in NZ to establish the benefits, or the environmental risks, associated with stocking of silver carp is lacking.

5.4.3 Constraints - *In what situations can the option not be applied?*

Silver carp are not an option where a high certainty of outcome for cyanobacteria control is required. The fish may selectively graze large-sized phytoplankton, but this could drive the phytoplankton community to be dominated by smaller species, which also can include problematic cyanobacteria (e.g. *Planktothrix*, *Cylindrospermopsis*). Equally, silver carp can consume large numbers of zooplankton and reduce the potential for zooplankton grazing pressure on smaller phytoplankton, resulting in an increase in phytoplankton biomass (Zhao et al 2013) and reduced water clarity. Impacts on fish fry that feed on zooplankton in surface waters are likely.

Rowe (2010) indicated the need for a moratorium on use in the catchment of the Waikato River, as there remains a question over whether the fish may breed in this river environment.

A requirement for containment of fish is required, as for grass carp.

Silver carp can also produce large amounts of floating faecal material under some conditions (Carruthers 1986), which may be aesthetically displeasing.

In the absence of specific information on silver carp requirements, considerations of the suitability of stormwater systems for fish survival should be considered similar to grass carp.

5.4.4 Requirements - *What other options / practices might be required in conjunction with this option?*

It has been suggested that silver carp should be stocked together with grass carp in artificial lakes (G. Jamieson NZ Waterways Restoration Ltd 2013 pers. comm.), however there is no information on the benefit arising from this and it is presumably a precautionary measure should cyanobacteria become dominant.

5.4.5 Track record - *Where has this option been successful / unsuccessful?*

Silver carp are successfully used in aquaculture ponds in the United States for water quality benefits (prevention of algal bloom and bust cycles) and to prevent algal impacts on the taste of aquaculture species. Rowe (2010) reviewed the use of silver carp in hyper-eutrophic water treatment facilities, but concluded results in terms of nutrient stripping and water quality benefits were variable. Stocking to lakes to improve water clarity also showed mixed results, however there was supporting evidence for reductions in cyanobacteria dominance and bloom prevention.

In NZ, trials of silver carp for the control of cyanobacteria blooms were carried out in Lake Orakai (4 ha), Hawkes Bay (Carruthers 1986), but there was little quantitative information on the extent, frequency or duration of control (Rowe 2010). Between 1989 and 1990, several hundred silver carp fry were released into Lake Omapere (Northland), but no monitoring of their fate or effects (beneficial and/or adverse) was undertaken.

5.4.6 Implementation - *Methods employed*

Required stocking rates of fish are expected to be higher in NZ than the warmer waters of countries where silver carp have been used, but optimal densities are not known. Stocking rates of 1500 to 3000 fish per ha were used in a eutrophic Hawkes Bay lake, at a size greater than 35 mm (Carruthers 1986).

Approvals required are as for grass carp. Due to the lack of information on silver carp, all applications are treated as experimental.

5.4.7 Operation, maintenance, monitoring and reporting

Silver carp use in stormwater systems should initially be accompanied by prior- and post-stocking monitoring for phytoplankton dynamics and other key water quality and ecological parameters, to establish the associated risk and benefits. Therefore costs for initial adoption are likely to be high. Maintenance and monitoring required, including maintenance of containment barriers, would be similar to grass carp.

5.4.8 Practical considerations *- e.g. social issues, access constraints, consent requirements, health and safety concerns*

See as for grass carp.

5.4.9 Financial costs

Costs are likely to be similar to grass carp and can only be indicated on a direct, per ha basis, excluding consultation, approvals and cost of screens.

Cost estimate per ha*

(i) Start-up / implementation (once-off): Low

(ii) Operation & maintenance (annual): Low

(iii) Monitoring (annual): Low

(iv) Decommissioning, if relevant (once-off): Low

(v) Overall cost over the lifetime of the intervention: High (25 year period)

*Costs do not include consultation, or approvals, which are likely to vary on a situational basis

5.5 Microbial control

5.5.1 Description and overview

Microbial control involves the introduction of bacteria to the water body, which then take up nitrogen and phosphorus, and therefore may out-compete algal populations. Some bacteria may also exert control on algal growth by excretion of lytic enzymes that break down algae cell walls (Rashidan and Bird 2001; Peng et al 2003; Choi et al 2005). Furthermore, studies have indicated that growth inhibition by bacteria may be higher for cyanobacteria compared with green algae and diatoms, thus there may be potential for selective control of cyanobacteria blooms (Ahn et al 2003). However, the efficacy of products *in situ* has been largely unexplored, as have unsupported claims that these products are free of unintended side effects (Schmack et al 2012).

5.5.2 Application - *In what situations can the option be applied?*

Applications are largely undocumented and there has not been rigorous scientific testing *in situ* to our knowledge. However, the limited data available suggests that they may be effective in controlling harmful cyanobacterial blooms in small systems (Rashidan and Bird 2001; Choi et al 2005; Schmack et al 2012). Trials of bacteria purported to take up nutrients have occurred in Lake Sullivan (Whakatane) where mesocosm experiments were undertaken, with inconsistent results; however, the trial was run over a very limited temporal and spatial scale, and a whole lake trial was recommended (Scholes 2005). A subsequent whole-lake trial was also inconclusive. A further trial was undertaken in a northern arm of Lake Rotoehu. No result has been reported from this trial and suggestions of improvements in water clarity should be carefully evaluated in light of the fact that small, deep arms of Lake Rotoehu generally have substantially lower levels of nutrients and chlorophyll, and higher clarity, than the central lake basin.

5.5.3 Constraints - *In what situations can the option not be applied?*

Largely unknown as not scientifically tested *in situ* to our knowledge.

5.5.4 Requirements - *What other options or practices might be required in conjunction with this option?*

This option would likely require effective monitoring both prior and after application to evaluate the efficacy of the product and potential consequences for the ecosystem. If whole-lake applications are considered then laboratory and **mesocosm** trials would be necessary at manufacturer-recommended whole lake concentrations, in order to carefully control multiple interacting variables.

5.5.5 Track record - *Where has this option been successful or unsuccessful?*

Microbial algal control has been trialled at a number of locations in Western Australia, e.g. Lake Bertram, Joondalup golf course, Perth. Anecdotal findings suggest algal growth was reduced but this has not been scientifically verified (Schmack et al 2012). Tank experiments using the same product indicated that the bacteria did not actively lyse algal cells, but nutrient concentrations were sometimes decreased in tanks with the bacteria (Schmack et al 2012). The authors' suggested, however, that the tank experiments were not ideally suited to evaluating the products' efficacy due to environmental conditions in the tank (particularly aeration) potentially affecting microbial growth. Other laboratory experiments have indicated that bacteria may have potential in controlling harmful algal blooms (Choi et al 2005).

5.5.6 Implementation - *Methods employed*

There is little available information on the precise methods or dose required with this option; however it would involve the introduction of the microbial products to water body (contained in wax blocks or in solution) in one or more applications.

5.5.7 Operation, maintenance, monitoring and reporting

Monitoring before and after the application would be useful to evaluate the benefits and effects, but if a whole lake application is undertaken then the evaluation should be cognisant of potential for rapid temporal changes in water quality that characterise many standing waters.

5.5.8 Practical considerations - *e.g. social issues, access constraints, consent requirements, health and safety concerns*

Introduction of a biological control agent, especially one that has not been extensively evaluated scientifically, may be negatively perceived by the general public. Approval may be required from relevant authorities to introduce a biological control agent into a waterbody.

5.5.9 Financial costs

Annual cost for treatment of Lake Sullivan was estimated to be c. $15,000 (Scholes 2005).

(i) Start up: Moderate
(ii) Operation and maintenance: Moderate
(iii) Monitoring: Recommended for evaluation of efficacy/effects - cost would be dependent on frequency and variables monitored
(iv) Decommissioning: NA
(v) Overall cost: Moderate-High

5.6 Barley straw

5.6.1 Description and overview

Barley straw has been used to control algal growth in ponds, lakes and reservoirs (Newman and Barrett 1993; Everall and Lees 1996; Ferrier et al 2005). Some studies suggest that barley straw can be effective as a treatment for algae blooms (Everall and Lees 1996; Everall and Lees 1997; Zheng et al 2006; Purcell et al 2012). The addition of barley straw does not typically kill algae already present (i.e. algaecidal effect), but may inhibit the growth of new algae (i.e. algaestatic effect) (Murray et al 2010). Other research indicates that the compounds released by barley straw are a selective inhibitor and so the treatment may alter the species composition of algae communities and algaecidal effects have been reported for certain species of phytoplankton (Ferrier et al 2005). Decomposition of the barley straw under aerobic conditions may produce the algaestatic (or algaecidal) compounds, but the straw needs to be removed before further decomposition releases nutrients that can stimulate algal growth (Drabkova 2007).

Natural inputs of terrestrial leaf litter may also provide significant inhibition of certain algal species, although efficacy is also dependent on (deciduous) tree species (Ridge et al 1999). This research was carried out in the UK and to our knowledge there has been no research into the potential for leaf litter from NZ trees to inhibit algal growth.

5.6.2 Application - *In what situations can the option be applied?*

The use of barley straw may be most effective in smaller systems and in systems where immediate or complete algae control is not required.

5.6.3 Constraints - *In what situations can the option not be applied?*

Barley straw is unlikely to be effective in very large systems, or those in which immediate control is required. It will also be ineffective in oxygen-poor water bodies, and may increase oxygen deficits in bottom waters of systems that are prone to **stratification**.

5.6.4 Requirements - *What other options or practices might be required in conjunction with this option?*

None known.

5.6.5 Track record - *Where has this option been successful or unsuccessful?*

Much of the research on barley straw has focussed on the control of cyanobacteria species. Barley straw has been used extensively in the United Kingdom in ponds, canals, streams and reservoirs. It has been successful in the control of *Microcystis* blooms, although there have been concurrent increases in green algae and diatoms (Purcell et al 2012). A study in NZ trialled the use of barley straw to control a green filamentous alga, *Hydrodictyon reticulatum*, water net (Wells et al 1999). Field and laboratory trials showed inconsistent effects of barley straw on water net growth, however, varying from inhibition to enhanced growth (Wells et al 1999).

5.6.6 Implementation - *Methods employed*

Barley straw must be used in aerobic conditions, so is usually deployed on the water surface, either in cages or netting bags that can be tethered to buoys or the side of a pond. Dosage varies in the literature, but positive results have been obtained with dosages of between 25 and 50 g m^{-3} (Everall and Lees 1996; Everall and Lees 1997). Application of excessive amounts (c. 500 g m^{-3}) should be avoided as the water may become deoxygenated (Newman 2004). In water at 10°C it takes c. 6 – 8 weeks for barley straw to become active, but only c. 1 – 2 weeks when water is 20°C (Newman 2004).

5.6.7 Operation, maintenance, monitoring and reporting

Decomposed straw needs to be removed (and replaced if required). The rate of decomposition will be dependent on water temperature, but will likely require removal and replacement relatively frequently (c. 4 - 6 months; Newman 2004; Drabkova 2007).

5.6.8 Practical considerations - *e.g. social issues, access constraints, consent requirements, health and safety concerns*

Access to the edge or middle of the pond will be required to tether the barley straw contained in cages/bags. As barley straw is a natural product, there are likely to be few social, or health and safety issues. However, if not maintained or removed properly, there may be aesthetic issues associated with decomposition (in addition to enhanced growth of algae due to nutrient release).

5.6.9 Financial costs

Cost will be dependent on size of lake, but is likely to be low for small ponds.

(i) Start up: Low

(ii) Operation and maintenance: Low

(iii) Monitoring: not required (but recommended for evaluation of efficacy/effects - cost would be dependent on frequency and variables monitored)

(iv) Decommissioning: Low

(v) Overall cost: Low

5.7 Macrophyte restoration

5.7.1 Description and overview

Macrophyte-dominated lakes and ponds are resistant to the development of algae blooms (Figure 7) because aquatic plants reduce wind-driven resuspension of bottom sediments, and because the macrophytes may take up nutrients that would otherwise be used to fuel algal growth (Scheffer 2005). Furthermore, some macrophytes may release compounds that inhibit algae (particularly cyanobacteria) growth (reviewed by Hu and Hong 2008). Restoration of macrophytes in ponds with high concentrations of algae and suspended sediments may be difficult, however, due to the elevated turbidity (Scheffer et al 1993). Thus, macrophyte restoration typically requires other control options (e.g. benthivorous fish removal) in place to reduce algae, suspended sediment and nutrient concentrations (Jeppesen et al 2012).

5.7.2 Application - In *what situations can the option be applied?*

Restoration of macrophyte dominance is likely to be successful only in shallow, sheltered water bodies, and where potential limiting factors have been controlled for (i.e. see "requirements" section below).

5.7.3 Constraints - In *what situations can the option not be applied?*

Macrophyte restoration to control algal growth is not applicable to deep lakes and ponds, with small littoral zones. It is also unlikely to be successful in very large, wind-exposed and shallow water bodies, due to the high rates of resuspension.

5.7.4 Requirements - *What other options or practices might be required in conjunction with this option?*

Restoration of macrophytes is likely to require a number of contemporaneous practices to increase the probability of success. Lowering turbidity by the use of flocculants (to control water column chlorophyll *a* and nutrient concentrations), and temporary or permanent installation of wave-breakers (to reduce resuspension) may be required. Reductions in benthivorous fish biomass and protection from waterfowl may also be necessary.

Figure 7 Macrophyte restoration in Lake Huizhou, China. Macrophyte (*Vallisneria* sp.) restored area on right side of the photo and the un-restored area on left side.

5.7.5 Track record - *Where has this option been successful or unsuccessful?*

A large-scale restoration experiment in Lake Wuli, China, used multiple restoration methods (including macrophyte planting, and fish removal) to reduce symptoms of eutrophication (Chen et al 2009). Macrophyte restoration was partially successful and there were decreases in nutrient concentration and algae biomass. It should be noted that the reduction in phytoplankton biomass lagged behind the improvement in water quality and macrophyte abundance. Thus, macrophyte restoration as a tool for algae control may not achieve results in the short-term (i.e. within 1 to 2 years), but may be effective longer-term. Rapid results have been obtained, however, in an experiment that used modified local soils mixed with macrophyte seeds to flocculate algae blooms and promote macrophyte restoration in a small (0.1 km^2), shallow (1.6 m) bay in Lake Tai, China (Pan et al 2011a). Macrophyte establishment and survival were found to be significantly enhanced in Lake Rotorua (Hamilton) when fish were excluded from shallow areas of the lake (de Winton et al 2001).

5.7.6 Implementation - *Methods employed*

Macrophyte seeds could be added to the littoral zone (usually mixed with suitable substrate) immediately following any other control methods (e.g. flocculants, fish removal). Transplanting of propagated cutting has also been used (Suren 2009). Both methods require that seeds or cuttings are screened for unwanted or exotic macrophyte species.

5.7.7 Operation, maintenance, monitoring and reporting

Re-seeding or re-planting may be required, depending on the success of the initial treatment. Macrophyte biomass may need monitoring to determine the efficacy of the option. In addition, if removal/reduction of benthivorous fish has been undertaken to allow macrophyte establishment then there may be on-going efforts required to maintain pest fish biomass at a low level.

5.7.8 Practical considerations - *e.g. social issues, access constraints, consent requirements, health and safety concerns*

Restoration of macrophytes may lead to elevated water levels in the pond or lake (if macrophyte biomass becomes very high), particularly in summer. Previous attempts to revegetate waterways with native macrophytes (and riparian plantings) have discovered that public perceptions of macrophyte and riparian growth can be negative due to perceived lack of maintenance and level of plant growth (Suren 2009). In addition, inadvertent restocking of lakes or ponds with benthivorous fish may also compromise restoration efforts (Robertson et al 2000). Thus, the continued success of this option may rely on a degree of community education and/or participation.

5.7.9 Financial costs

Financial costs are likely to be highly variable, depending on the number and scale of contemporaneous practices required.

(i) Start up: Moderate-High
(ii) Operation and maintenance: Low-Moderate
(iii) Monitoring: (recommended for evaluation of efficacy/effects) Low
(iv) Decommissioning: NA
(v) Overall cost: Moderate-High

5.8 Pest fish removal

5.8.1 Description and overview

Some pest fish species, such as koi carp, significantly disturb sediment and uproot macrophytes when they feed, which increases water column sediment and nutrient concentrations, and enhances phytoplankton and reduces macrophyte biomass (de Winton et al 2001; Rowe 2007). Removal of pest fish from a water body can reduce suspended sediment, nutrient and algae concentrations. Introduction of coarse fish into waterbodies (e.g. for aesthetic, cultural or fishing purposes) is both illegal and strongly discouraged. Removal methods for koi carp include one-way barriers that stop migration into a water body but allow the fish to leave, electro-fishing, netting and trapping (Gilligan et al 2005). In addition, chemical control (e.g. with rotenone) can be effective in small water bodies, although this requires careful consideration due to the effect of the chemicals on non-target species (Ling 2003). Pest fish removal can also potentially release control on zooplankton, allowing for increased zooplankton populations that may then exert top-down control on phytoplankton (see 5.9; Zooplankton and invertebrates).

5.8.2 Application - In *what situations can the option be applied?*

Applicability will depend to some degree on the method of pest fish removal/control used. One-way barriers may be effective (even in larger water bodies) if pest fish can only re-invade the lake from inlets or outlets that can be controlled. Electro-fishing, trapping or chemical control may be effective in smaller lakes that are not highly connected to water bodies with pest fish populations (Hicks et al 2008; Daniel and Morgan 2011).

5.8.3 Constraints - In *what situations can the option not be applied?*

This option is unlikely to be applicable in systems that are connected to other water bodies with pest fish populations and the potential for re-invasion cannot be minimised (e.g. with one-way barriers).

5.8.4 Requirements - *What other options or practices might be required in conjunction with this option?*

Electro-fishing and netting can be more effective if the water level is lowered prior to attempting removal of fish (Hicks et al 2008) and rotenone can be more effective if macrophytes are removed as dense macrophyte stands can prevent mixing of rotenone and provide refugia for the fish (Rowe 2001). Consideration may also need to be given to both surface and submerged inflows that may provide refugia for fish while areas of the main basin are being treated. Thus effective lake rotenone operations have included treatment of inflows (Robertson et al 2000).

5.8.5 Track record - *Where has this option been successful or unsuccessful?*

Monofilament gill nets were used in the Rotopiko lakes (Waikato) to control rudd populations with some degree of success (Neilson et al 2004). However post-treatment sampling showed that, even following intensive removal, pest fish remained in the lakes. Other methods typically have been unable to achieve 100% removal of target species. For example, boat electro-fishing and gill netting were used in ornamental ponds at Kauri Point (Katikati). Although, c. 80% of the koi population were estimated to have been removed after 2 days of effort, poor water visibility and the presence of koi carp breeding in the ponds prevented complete eradication (Hicks et al 2008). In Lake Ohinewai (Waikato), a combination of removal methods including electro-fishing, baited traps and fyke nets were able to remove c. 60 % of the pest fish (koi carp, catfish and goldfish) (Daniel and Morgan 2011).

5.8.6 Implementation - *Methods employed*

Methodology depends on the method of removal. Electro-fishing requires a specially-equipped boat; various methods are available for netting and trapping, and rotenone is available as crystalline preparations, emulsified solutions, or as dust (Ling 2003). However, use of rotenone, at the present time, is restricted to Department of Conservation and usually requires special consideration and approvals.

5.8.7 Operation, maintenance, monitoring and reporting

As most methods have not been able to remove 100% of target species, monitoring and re-treatment (e.g. by further fishing, netting) may be required.

5.8.8 Practical considerations - ***e.g. social issues, access constraints, consent requirements, health and safety concerns***

Approvals and permits required will be dependent on the method of fish removal used and the owner of the property on which the pond or lake is situated. Some methods may be subject to approval from agencies such as the Department of Conservation. Application of rotenone will require a license.

5.8.9 Financial costs

Financial costs are likely to be variable, depending on the method of removal used. However, for the majority of methods the total cost is likely to be high.

(i) Start up: Moderate-high
(ii) Operation and maintenance: Moderate-high
(iii) Monitoring: Moderate-high
(iv) Decommissioning: NA-Low
(v) Overall cost: High

5.9 Zooplankton and invertebrates

5.9.1 Description and overview

Zooplankton or filter-feeding/grazing invertebrates may be able to provide a control on algal biomass in lakes and ponds (Dionisio Pires et al 2005; Sierp et al 2009). Biomanipulation of the trophic cascade (e.g. by removing zooplanktivorous fishes to increase zooplankton biomass) may be effective, particularly in small water bodies (Hosper and Jagtman 1990; Meijer et al 1999; Van de Bund and Van Donk 2002). This method is not always successful however (Noonan 1998), and grazing of phytoplankton by zooplankton may be limited when (typically) inedible cyanobacteria are the dominant phytoplankton species (Chen et al 2009).

The use of mussels or other filter-feeding bivalves, such as clams, may be an alternative means to control phytoplankton (Dionisio Pires et al 2005). Filter-feeding bivalves have long been described as a "natural eutrophication control" as they can be capable of substantial top-down control on phytoplankton populations (Officer et al 1982; Smaal and Prins 1993; Dame and Prins 1998). However, the ability of a population to exert significant control is system- and species-specific, dependent on a number of factors, including bivalve clearance time, water residence time and phytoplankton production time (Dame and Prins 1998). In NZ, the native freshwater bivalve mostly likely to exert some control on phytoplankton is the freshwater mussel (kakahi), the most common species of which is *Echyridella menziesii* (Ogilvie and Mitchell 1995). However, the life-cycle of these mussels includes a larval stage on a host fish, typically native species such as eels or koaro (although trout may also be a suitable host). There are substantial knowledge gaps regarding the suitability and implementation of native freshwater mussels for biomanipulation in NZ, which would need to be addressed to assess the suitability of this option (Phillips 2007). Introduction of exotic bivalve species to ponds and lakes is not recommended as they are likely to become invasive species, e.g. zebra mussels (*Dreissena polymorpha*) in North America and the UK, which can negatively impact on native species and ecosystem functioning.

Finally, abundant populations of invertebrate grazers, such as snails, may control periphyton growth on macrophytes, enhancing macrophyte growth and stabilising the clear-water state; thus biomanipulation of lake or pond food webs that enhance snail biomass may also control algae populations (Lombardo 2005).

5.9.2 Application - *In what situations can the option be applied?*

Biomanipulation to increase zooplankton biomass can be applied in water bodies that contain zooplanktivorous fish and where removal of those fish is possible (see 5.8; Pest fish removal). Biomanipulation using freshwater mussels may be effective in shallow ponds or lakes with long water residence times, where it is possible that total filtration rates may be comparable to phytoplankton growth rates *in situ* (Ogilvie and Mitchell 1995).

5.9.3 Constraints - *In what situations can the option not be applied?*

Algae control by zooplankton or filter-feeding or grazing invertebrates is unlikely to be very effective in deep ponds or lakes. Re-establishment of freshwater mussels is likely contingent on suitable substrate (sandy sediment), sufficient oxygen at the sediment-water interface and presence of fish that provide a suitable host for the larval stage.

5.9.4 Requirements - *What other options or practices might be required in conjunction with this option?*

To increase zooplankton biomass, removal of zooplanktivorous fish would be desirable. Habitat restoration for native fish species would likely assist with biomanipulation using freshwater mussels.

5.9.5 Track record - *Where has this option been successful or unsuccessful?*

The removal of all fish from Lake Parkinson (near Auckland) using rotenone resulted in improved water clarity, thought to be due to increased zooplankton populations exerting a top-down control on phytoplankton (Rowe 2007). It should be noted, however, that it is typically the juvenile stage of exotic fish species in NZ that are planktivorous. Therefore fish removal methods that do not target juveniles as well as adults may not be effective in this regard.

The successful control of phytoplankton populations by bivalves has typically been observed under natural conditions or as a result of an invasion, i.e. not as a result of an intentional biomanipulation. A scoping study for possible biomanipulation of ornamental ponds in Washington, DC, found that the option was unlikely to be successful due to the high biomass of mussels required exert significant control (Phelps 2005). Similar calculations for Lake Rotorua

(Bay of Plenty) and Lake Tuakitoto (Otago) indicated that the density of mussels required typically exceeded that of naturally recorded densities (Phillips 2007).

Gastropods (i.e. snails) have been shown to be an effective control on a filamentous algae (water net, *Hydrodictyon reticulatum*) in Lake Aniwhenua, NZ (Wells and Clayton 2001).

5.9.6 Implementation - *Methods employed*

For removal of some zooplanktivorous fishes see 5.8; Pest fish removal. However, there are also native fish species that may be planktivorous for all or part of their life-cycle, and therefore there may be significant top-down control of zooplankton in natural water bodies that have not previously contained pest-fish and/or been subject to pest-fish removal attempts (Jeppesen et al 1997).

Further research is likely required to guide implementation of biomanipulation with freshwater mussels (Phillips 2007).

Densities of gastropods can be most easily manipulated in small ponds, and will typically be better protected from fish predators if there are macrophyte beds present (Lombardo 2005).

5.9.7 Operation, maintenance, monitoring and reporting

For all biomanipulation options short to medium term monitoring (of water quality and the relevant biomanipulation agent) will likely be required to assess the efficacy of the intervention. Furthermore, effective monitoring will be useful to determine what, if any, further intervention is required to maintain zooplankton, bivalve or snail populations in the water body.

5.9.8 Practical considerations - *e.g. social issues, access constraints, consent requirements, health and safety concerns*

As these biomanipulation options typically involve removing invasive or exotic species and enhancing native species diversity, they may be attractive options within the wider community. A notable exception may be removal of coarse fish from ponds that are used by members of the public for recreational coarse fishing.

Introduction of a species to a waterway (that has not previously been known to exist there) will require consent from the Department of Conservation.

5.9.9 Financial costs

Financial costs will be dependent on the biomanipulation method employed, and are likely to be highly variable. However, biomanipulation is typically seen as a low cost option, relative to controls that aim to reduce external loading of nutrients and sediment.

(i) Start up: highly variable
(ii) Operation and maintenance: highly variable
(iii) Monitoring: highly variable
(iv) Decommissioning: highly variable
(v) Overall cost: highly variable

5.10 Waterfowl management

5.10.1 Description and overview

Increased nutrient concentrations and enhanced algal growth can result where high numbers of waterfowl are present on stormwater systems. Urban waterfowl such as geese and ducks provide large inputs of faecal matter (Figure 8) or disturb sediment during feeding. In contrast, it is possible that some birds (i.e. shags) may exert some top-down control on pest fish species in some NZ ponds and lakes, potentially improving water quality. However, research is required to test this theory and to determine the net effect of shags on these systems (Rowe et al 2008).

Figure 8 Damaged willow trees at Lake Johnson (Otago) where shags roost. Note algal bloom.

There is evidence to suggest that waterfowl may prefer ponds with short (i.e. mown) vegetation; thus allowing more natural vegetation to grow around the margins of ponds may reduce waterfowl use of these areas (Smith 2006). Bird use of stormwater ponds has also been shown to be correlated with pond surface area, the ratio of open water area to the area of emergent or woody vegetation, perimeter irregularity, and geographic isolation (Blackwell et al 2008). However, there was variability in the strength and direction of the relationships, dependent on bird species. In general, the authors concluded that to reduce bird use of stormwater ponds the pond perimeter should be minimised with circular or linear designs, and access to open water reduced by use of a cover or water drawdown (Blackwell et al 2008).

Ponds with islands may be attractive as nesting habitat for some waterfowl species, and waterfowl may also be attracted to ponds where their feed is supplemented by humans, so cessation of feeding will likely reduce waterfowl numbers. Waterfowl can be deterred from using ponds with predator decoys, or they may be trapped and moved away from the area by a wildlife control operator.

5.10.2 Application - In *what situations can the option be applied?*

There are options for waterfowl management that are likely to be applicable to most situations. Reducing mowing of grass around pond perimeters and allowing more natural vegetation to grow may be applicable where pond design and surroundings allow. For ponds that are accessible to the general public, feeding of waterfowl can be discouraged.

5.10.3 Constraints - In *what situations can the option not be applied?*

Reducing pond area by water level drawdown may not be appropriate as the purpose of stormwater ponds is for retention of water. Covers are unlikely to be practical or cost-effective for large ponds.

5.10.4 Requirements - *What other options or practices might be required in conjunction with this option?*

None known.

5.10.5 Track record - *Where has this option been successful or unsuccessful?*

A rotating, intermittent beacon, and a gas powered sonic gun, were found to be a successful deterrents to waterfowl landing on toxic ponds in Western Australia (Read 1999). These types of deterrents are unlikely to be applicable in urban settings, however. Evidence for other waterfowl management measures, e.g. reducing mown areas, is largely anecdotal; a review of available literature has not shown any published studies that determine the efficacy of these options.

5.10.6 Implementation - *Methods employed*

Discouraging feeding by people will likely require a degree of education to stimulate public support and participation. Signs can be posted in public areas that say "do not feed the ducks". Planting with native trees and shrubs will be required to reduce mown grassy areas.

5.10.7 Operation, maintenance, monitoring and reporting

Monitoring waterfowl use of ponds in the long term will be necessary to determine the need for initial and further control. Appropriate plantings and signs will require little maintenance.

5.10.8 Practical considerations - *e.g. social issues, access constraints, consent requirements, health and safety concerns*

Trapping and removal of waterfowl may require approval from, or consultation with, various agencies such as Fish and Game NZ, and/or Department of Conservation.

5.10.9 Financial costs

For most of the options to do with waterfowl management the costs are likely to be low (with the exception of trapping and relocating large numbers of birds).

(i) Start up: Low
(ii) Operation and maintenance: Low
(iii) Monitoring: Low
(iv) Decommissioning: NA
(v) Overall cost: Low

Chemical control

Chemical control options for both plants and algae

5.11 Chelated Copper

5.11.1 Description and overview

Algaecides (e.g. copper sulphate, copper chelates, endothall amine are chemical substances that rely on direct contact with algae to exert toxic effects on biochemical processes in the algal cells. Copper-based algaecides are commonly used in swimming pools, aquaria and in many natural water bodies to control algae. A solution of pure copper sulphate would be highly toxic to other aquatic organisms as well as algae. To reduce toxicity, the copper is typically chelated, i.e. the copper ion is bonded to other compounds and so is less reactive with organisms and organic matter (Clearwater et al 2007a). Dissolved copper toxicity is significantly affected by water pH, hardness and dissolved organic matter (DOM). Toxicity is reduced in hard, high pH waters and also in waters with high concentrations of DOM, which binds copper and decreases bioavailability and toxicity. However, neither copper sulphate, nor chelated copper are registered for use in natural waters in NZ.

There may also be disadvantages in using copper as a treatment for cyanobacterial blooms, as cyanobacteria toxins can be released into the water from dying cells following treatment (Jančula and Maršálek 2011). In a well-known example, 148 people required hospitalisation with symptoms of gastro-enteritis following dosing of a local water supply on Palm Island (Australia) with copper sulphate to control an algae bloom (Hawkins et al 1985). Furthermore, repeated use can result in accumulation of copper in sediments, potentially restricting sediment removal and disposal (Drabkova 2007). Copper applications may alleviate common symptoms of water quality problems such as algal blooms but do not generally address underlying causes such as excess nutrients; lysing of cells releases nutrients into the water column that are then available when the toxic effect of copper subsides.

5.11.2 Application - *In what situations can the option be applied?*

None (other than for biosecurity emergencies, such as a didymo incursion in the North Island).

5.11.3 Constraints - *In what situations can the option not be applied?*

Cannot be used in waterways (without special provisions) as it is not registered for use in NZ.

5.11.4 Requirements - *What other options / practices might be required in conjunction with this option?*

Option would be stand alone.

5.11.5 Track record - *Where has this option been successful / unsuccessful?*

Copper has been used for more than 60 years in reservoirs in the USA. In NZ it has been used in natural waterways in the past but is no longer used in this way.

Chelated copper was used to control water net in Lake Aniwhenua, successfully killing surface mats and mid-water growths, but did not kill algae growing on the bottom (Wells et al 1999).

More recently GemexTM was trialled extensively for didymo control by NIWA (Clearwater et al 2007b) culminating in a field trial run at Princhester Creek, Central Otago. It had good algicidal properties but did not eradicate the alga in this trial. It was proposed for use in the Takaka River for didymo control but a NIWA report (Wells et al 2007) did not support its use.

Copper sulphate has been used in Modellers Pond, Nelson (a saline pond that receives stormwater runoff) to control algae growth, but is now prohibited as copper has built up in the sediments to levels exceeding ANZECC guidelines (Wells et al 2010).

5.11.6 Implementation - *Methods employed*

Chelated copper would be applied in a similar way to herbicides, to achieve a required concentration-contact time. Use would be for biosecurity emergencies only and likely to have specific use requirements.

5.11.7 Operation, maintenance, monitoring and reporting

Use would be for biosecurity emergencies only and under strict conditions, monitoring and reporting.

5.11.8 Practical considerations *- e.g. social issues, access constraints, consent requirements, health and safety concerns*

Copper dosing has resulted in high levels of this heavy metal accumulating in the sediments (Cooke et al 1993; Wells et al 2010) and concerns have been raised regarding long term use. A number of summaries exist on the impact of copper in aquatic ecosystems (Demayo et al 1982; McKnight et al 1983) and its use in natural water ways has been reduced markedly or withdrawn overseas.

5.11.9 Financial costs

Likely to be similar to other herbicides.

Cost estimate per ha

(i) Start-up / implementation (once-off): Moderate
(ii) Operation & maintenance (annual): Low
(iii) Monitoring: Low
(iv) Decommissioning, if relevant (once-off): NA
(v) Overall cost over the lifetime of the intervention (maximum of 25 years): Moderate

Chemical control options for plants

5.12 Glyphosate isopropylamine

5.12.1 Description and overview

Glyphosate isopropylamine is a broad-spectrum, non-selective, systemic herbicide that works by inhibiting protein synthesis in plants. When applied to green tissue, it is translocated to growing points, including below ground organs and is effective against a wide range of plants on land or emerging from the water.

A number of marketed products have glyphosate as the active ingredient (a.i.) and these may be augmented by surfactants and adjuvants. Only products labelled for use around waterways should be used where contamination of water may occur, due to the toxicity of some types of surfactants for aquatic life. Formulations generally have 360 g per L glyphosate isopropylamine as a soluble concentrate.

5.12.2 Application - *In what situations can the option be applied?*

Glyphosate should be applied to actively growing target plants and is effective against emergent and marginal plants and trees such as willows. This herbicide would be well suited to where a blanket control is required e.g. emergency spillways, embankment dams.

5.12.3 Constraints - *In what situations can the option not be applied?*

Glyphosate isopropylamine does not affect submerged aquatic plants and does not **adequately control** alligator weed, Manchurian wild rice, phragmites, purple loosestrife, sagittaria, Senegal tea or spartina. It is less effective against rhizomatous species and, as it is non-selective, it can easily damage non target plants. Effectiveness can be reduced by rainfall within a few hours of application. Efficacy is reduced in stressed plants (e.g. wilting) and where plant surfaces are dirty.

5.12.4 Requirements - *What other options / practices might be required in conjunction with this option?*

Mowing may be used ahead of treatment time to produce new growth more amenable to herbicide coverage and translocation. Post-treatment burning to remove dead biomass of marginal emergent plants is not advised in suburban environments.

5.12.5 Track record - *Where has this option been successful / unsuccessful?*

In NZ has been used to manage crack (*Salix* x *fragilis*) and grey willow both aerially and via drill and inject. Effective in control of grasses (including Mercer grass, kikuyu, pampas, tall fescue, glyceria, reed canary grass, creeping bent) also sedges (e.g. rautahi), some rushes, floating spp (salvinia, water hyacinth), floating leaved (water poppy, water lilies), raupo, willow weeds, water cress etc.

5.12.6 Implementation - *Methods employed*

Use of herbicides should be always be guided by label information and/or manufacturer's directions.

Glyphosate may be sprayed or wiped onto green plant surfaces, woody targets may be drilled and injected or stumps painted with the herbicide. Non-target impacts are minimised by careful application. At higher levels of application a spray mix of 8.1 g per L (or mg per kg) should be applied at the rate of 9 L of the 360g per L a.i. applied per hectare.

Application should seek to reduce environmental loads by treating before weed seed-set and spraying banks when water levels are low.

5.12.7 Operation, maintenance, monitoring and reporting

It takes several weeks for susceptible plants to die off, and may need follow-up where germination of plants occurs throughout the growing season e.g. willow weeds. Monitoring is required for the best timing of treatment, and to determine the period before re-treatment is required.

5.12.8 Practical considerations *- e.g. social issues, access constraints, consent requirements, health and safety concerns*

In most instances, the use of this herbicide in these environments is not subject to resource consent requirements.

Glyphosate use is widespread in NZ. It does not bioaccumulate, biomagnify, or persist in a biologically available form in the environment and, as the mechanism of action is specific to plants, it is relatively nontoxic to animals (Solomon and Thompson 2003). In most situations glyphosate is inactivated on contact with soil and has no residual activity

5.12.9 Financial costs

Product costs for glyphosate are approximately $45 per ha. At an assumption of 1-2 applications per year and application costs of $100 per ha (costs variable, depending on application method), annual costs are likely to be $290 per annum.

Cost estimate per ha

(i) Start-up / implementation (once-off): Nil
(ii) Operation & maintenance (annual): Low
(iii) Monitoring: Low
(iv) Decommissioning, if relevant (once-off): NA
(v) Overall cost over the lifetime of the intervention (maximum of 25 years): Low

5.13 Diquat

5.13.1 Description and overview

Diquat is a quick acting contact herbicide, which has minimal translocation within plants. Mode of action is interruption of the electron transport system in plant photosynthesis, resulting in the formation of hydrogen peroxide, which then desiccates green plant tissue.

Diquat is one of two herbicides that are registered for use in water in NZ, and is used in the control of submerged weeds. It is available as either aqueous or gel formulations, the latter to assist in placement within the water column of deeper lakes (i.e. sink through thermal density gradients). Aqueous products (e.g. Reglone®) have 20% diquat dibromide (the a.i.) and require 1 mg per litre of the a.i. to control weeds (i.e. a 100,000 × dilution). Generic diquat products are available with the same a.i. ingredient at a cheaper price, but absence of additives such as colour, **emetic** and **stenching** additives should be considered from a handling and safety perspective.

Diquat has a selective action, controlling most of the major submerged pest species found in Auckland region (e.g. egeria, hornwort, lagarosiphon, elodea), yet native plant species are either largely unaffected (emergent plants, charophytes) or little affected (pondweeds, milfoils) compared to targeted weed species. This herbicide may greatly reduce the biomass of submerged weeds, but is unlikely to totally eradicate target species. Control is usually achieved for a season, or up to 1 year from treatment.

5.13.2 Application - *In what situations can the option be applied?*

Diquat can be used under static water conditions to reduce the height and size of weed beds. Efficacy depends upon the contact time achieved for an effective concentration of the herbicide in the weed bed, and performance is generally better in dense weed beds than in open beds.

5.13.3 Constraints - *In what situations can the option not be applied?*

Diquat is not so effective in situations where flowing water dilutes the herbicide and reduces the contact time. Diquat efficacy is generally reduced in turbid water (Hofstra et al 2001) or where plants are covered in organic matter or deposits of silt, which can rapidly bind up the diquat.

Diquat does not control eel grass.

5.13.4 Requirements - *What other options / practices might be required in conjunction with this option?*

Diquat might be used to reduce weed biomass prior to bottom lining, or to reduce required stocking densities of grass carp.

5.13.5 Track record - *Where has this option been successful / unsuccessful?*

Diquat has a long (over 50 years) proven history of use in NZ. Hornwort was reduced to about 5% of its original abundance within 6 weeks of treatment by diquat over 6 ha in Lake Wiritoa (Manawatu-Wanganui) with consequential increases in native vegetation cover (Wells and Clayton 2005). Similar results for other target species have been evident from use in the Rotorua lakes, and Lake Rotoroa (Hamilton Lake).

5.13.6 Implementation - *Methods employed*

Treatment should follow herbicide label requirements.

Aerial application in systems with a high weed biomass to water volume ratio should restrict treatments to 25% of the open water area at a time to avoid reduced dissolved oxygen effects on the system from decaying plants.

The product label for Reglone® states the maximum application rate of 30 litres per ha, regardless of water depth. A water depth greater than 0.5 m would act to dilute applied diquat to <1 mg per litre (Clayton and Severne 2006), but weed control has been achieved with application through several metres of depth of water. Therefore, in small, shallow water systems the natural dispersion of diquat is likely to result in more widespread herbicidal action against target species than just in the treated area, application can be conservative in use of product, and a single treatment of <25% of the open water area may be sufficient to achieve widespread control.

Surface application of diquat by hand sprayer is suitable for small systems (Figure 9). Larger systems can be treated by boat using surface booms or subsurface injection via trailing hoses. Helicopter application is not suitable for small systems due to the risk of drift and over spray.

An assessment of suitability of water quality and plant condition (cleanliness of target plants) is advised before spraying.

Large quantities of decaying weed can reduce dissolved oxygen levels in water. For this reason application may be best timed for spring or autumn. However, low dissolved oxygen levels also occur in dense weed beds overnight, and so treatment may ultimately increase dissolved oxygen levels (Clayton and Severne 2006). Likewise weed control activities should be avoided at times when cases of **avian botulism** may develop (e.g. water temperatures over 25°C), as the addition of decaying weed material at these times can exacerbate outbreaks.

Figure 9 Hand application of diquat to a small aquatic system (Photo: R. Wells, NIWA).

5.13.7 Operation, maintenance, monitoring and reporting

A resource consent may be required for the application of diquat, however it is also a permitted activity in many regions. Public notification (paper, radio and signs) would be required if areas to be treated are accessible to the public. Pre- and post-application assessment of weed abundance should be made to identify the level of control achieved. If fish or other wildlife are known to be present then monitoring should consider the potential for any mortality arising from deoxygenation.

5.13.8 Practical considerations *- e.g. social issues, access constraints, consent requirements, health and safety concerns*

Diquat in undiluted form is a Class 3 poison and is therefore a hazardous substance. However, when diluted to treatment concentrations diquat has very low risk to human and aquatic biota (Clayton and Severne 2006). When applied, diquat is rapidly absorbed by submerged aquatic plants and both inorganic and organic compounds within the water and bottom sediments. This means it is available in the water column for a very short timeframe (minutes to hours). Adsorbed diquat has no residual toxicity, is not biologically active and is degraded slowly by microbial organisms within sediments. No accumulation of diquat was detected in sediment at sites that have been regularly treated (HortResearch 2001).

Removal of the influence of submerged plants by herbicide in shallow ponds may result in concomitant greater algal biomass (Wakelin et al 2003).

5.13.9 Financial costs

Product and application costs are approximately $1600 per hectare. Assuming 25% of system treated, costs would be $400 per ha, per annum.

Cost estimate per ha*

(i) Start-up / implementation (once-off): Low.

(ii) Operation & maintenance (annual): Low

(iii) Monitoring: Low

(iv) Decommissioning, if relevant (once-off): NA

(v) Overall cost over the lifetime of the intervention (maximum of 25 years): Moderate

* Costs do not include approvals, consent costs, public notifications, compliance monitoring, or reporting, which are likely to vary on a situational basis.

5.14 Endothall

5.14.1 Description and overview

Endothall is one of two herbicides registered for aquatic use in NZ. The a.i. is the dipotassium salt of endothall and it is marketed as two formulations; Aquathol® K (aqueous) and Aquathol® Super K (pellets). Endothall is contact herbicide that affects protein synthesis and auxin production (plant growth hormone).

Endothall is used for submerged aquatic weed control. Appropriate use can result in one season control, to one year duration of control against target species. Eradication of the target species is possible under some circumstances.

5.14.2 Application *- In what situations can the option be applied?*

Endothall is effective on submerged aquatic weed species such as hydrilla, lagarosiphon, hornwort, and sometimes on parrots feather and pondweeds, but does not harm many native aquatic plants. Endothall has shown potential to selectively eradicate target pest plants such as hornwort and lagarosiphon, while leaving most native plants untouched. Native milfoils and pondweeds are able to recover from seed bank or buried rhizomes, while vegetative producing targeted weed species are less able to recover in the absence of any seed back.

Endothall is not affected by turbid water which can pose a constraint on the use of diquat.

5.14.3 Constraints *- In what situations can the option not be applied?*

Endothall is not effective on egeria, elodea, or eel grass.

Endothall cannot be used (EPA condition) in estuarine areas or water bodies within 1 km of the coast between 1 May – 31 August inclusive. Also the following timing restrictions apply:

- Where practicable, application of the substances within 250 m of any school shall be carried out during the school holidays.
- Where practicable, application of the substances shall not occur on weekends or during public holidays, including the period between 20 December and 10 January (inclusive).

Label restrictions relating to swimming, water takes for irrigation and water supply, and fish consumption exist (of lesser importance for stormwater systems). The public should be notified of the operation, excluded during treatments, and informed of risks.

Similar considerations of oxygen depletion and outbreaks of avian botulism exist as for diquat treatments (5.13; Diquat).

5.14.4 Requirements - *What other options / practices might be required in conjunction with this option?*

Consideration of appropriate formulation (pellets or aqueous) may be required. If eradication of a submersed weed species is desired, then treatment may require both if the target species grows amongst emergent species and in open water.

5.14.5 Track record - *Where has this option been successful / unsuccessful?*

Endothall has a proven record as an effective aquatic plant management tool in the USA (Sprecher et al 2002) particularly for hydrilla (*Hydrilla verticillata* (L. f.) Royle) and Eurasian watermilfoil (*Myriophyllum spicatum* L.) (Sisneros et al 1998; Netherland et al 1991; Parsons et al 2004).

In NZ endothall was used to eradicate hornwort from Centennial Lake, Timaru, with one treatment of endothall at 5 mg per L (Wells and Champion 2010). Similarly, lagarosiphon appears to have been eradicated in five out of six water bodies (up to 3.9 ha) four years after treatment with endothall concentrations as low as 0.11 mg per L under cool (~16ºC) Southland temperatures (Wells and Champion 2010).

In situations with flowing water results have been poor, with the exception of where it has been applied using a “drip feed” delivery system to maintain an adequate contact time. It must be in contact with the pest plant for sufficient length of time to give an effective herbicidal dose.

5.14.6 Implementation - *Methods employed*

Always follow label recommendations when handling or applying the product. Recommended application rates for a system 1 m deep are 60 - 98 L per ha (3 – 5 mg per L) for aqueous, and 48 – 80 kg per ha in 1 m deep (for 3 – 5 mg per L) for pellets. Effective herbicidal dose depends on concentration and contact time. As an indication it requires about 1 hour at 3 ppm or longer at lower concentrations (for example > 1 week at 0.01 ppm).

Surface application of endothall by hand sprayer or gun and hose is suitable for small systems. Water does not require being sprayed evenly (as required in a terrestrial situation) and so a D3 or D4 nozzle is best as it will not produce fine droplets that can drift from the application area. Larger application areas can be treated by boat using subsurface injection via trailing hoses. Helicopter application is not suitable for small systems due to the risk of drift and over spray.

The pellets can be applied using a pellet thrower or cast by hand.

5.14.7 Operation, maintenance, monitoring and reporting

Monitoring and reporting are subject to Resource Consent and EPA requirements. EPA requires a permission holder to undertake monitoring of: pH, temperature, dissolved oxygen, percentage pest plant cover and other native plants; at least 5 days prior to application, and 15-20 days after application.

5.14.8 Practical considerations *- e.g. social issues, access constraints, consent requirements, health and safety concerns*

Note Aquathol concentrate (as sold in containers with 40% endothall dipotassium salt) is hazardous. The concentrate is in the order of 100,000 times or more concentrated than the herbicidal rate. The label states 'do not allow any persons entry (without personal protective equipment) to the operational area while treatment is in progress'. An additional precaution is the restriction on swimming for 24 hours after application.

Once applied, endothall is rapidly diluted and breaks down to simple elements carbon, hydrogen and oxygen, and organic acids, with rates of breakdown speeded by microbial degradation and slowed by cool temperatures. There are no metabolites or breakdown products that are harmful in the environment. Typical half-lives are 3 – 7 days at 20°C.

As long as the label instructions and EPA requirements are met, health and safety concerns are minimal during application, and are negligible following application.

5.14.9 Financial costs

Upper costs for product is $3494 per ha for pellets. At an assumption of 1-2 applications for an eradication attempt and application costs of $100 per ha, annual costs are likely to be $7000.

Cost estimate per ha*

(i) Start-up / implementation (once-off): Moderate

(ii) Operation & maintenance (annual): Low

(iii) Monitoring: Low

(iv) Decommissioning, if relevant (once-off): NA

(v) Overall cost over the lifetime of the intervention (maximum of 25 years): Moderate

* Costs do not include approvals, consent costs, public notifications, compliance monitoring, or reporting, which are likely to vary on a situational basis.

5.15 Restricted herbicides

5.15.1 Description and overview

Four herbicides (metsulfuron methyl, haloxyfop methyl, imazapyr isopropylamine and triclopyr triethylamine (TEA)) have been approved by the NZ Environmental Protection Agency (EPA) for restricted use over water, by authorised agencies, under a set of conditions. The purpose of this restricted registration is to enable the eradication and control of priority national and regional pest plants which are otherwise difficult to control. Therefore these chemicals are only likely to be employed in stormwater systems by AC staff for species-specific control strategies. This is likely to be for 'Total Control' species under the RPMS (Table 1).

The following information was taken from a review by Champion (2012a) and evidence presented during the EPA application approval process (Champion 2012b and c).

5.15.2 Application - *In what situations can the option be applied?*

Triclopyr TEA is reported to control alligator weed, parrot's feather, willows, purple loosestrife, and water hyacinth. It is selectively highly toxic to a range of predominantly dicotyledonous plants, but grasses and sedges are unharmed.

Metsulfuron methyl provides control of alligator weed, yellow flag and arrowhead. It is rapidly taken up by plant roots and foliage, is translocated throughout the plant, but is not persistent. In tolerant plants (e.g. grasses and sedges), this chemical is broken down to non-herbicidal products.

Haloxyfop methyl to control pest grasses including spartina, Manchurian wild rice and saltwater paspalum (*Paspalum vaginatum*), but at the same rates it does not appear to affect non-grasses including sedges, rushes and other monocotyledons and dicotyledons.

Imazapyr isopropylamine is reported to control marshwort, fringed waterlily, yellow waterlily, water hyacinth, alligator weed, arrowhead, sagittaria, spartina, giant reed, phragmites, purple loosestrife and willows. This herbicide is toxic to a wide range of plant species.

5.15.3 Constraints - *In what situations can the option not be applied?*

These herbicides are only approved for use over water by authorised agencies including AC. A number of conditions (not provided here) are required for use. These products would only be used by organisations with biosecurity responsibilities (e.g. AC, DOC, MPI) and are not permitted for use for other purposes.

5.15.4 Requirements - *What other options / practices might be required in conjunction with this option?*

All applications of these herbicides require a resource consent from AC that must comply with conditions set by EPA.

5.15.5 Track record - *Where has this option been successful / unsuccessful?*

Triclopyr TEA and imazapyr isopropylamine are registered for aquatic plant control in the USA. In NZ, Triclopyr TEA is effective on purple loosestrife, water celery, water cress, primrose willow, and has successfully eradicated parrots feather. Imazapyr eradicated phragmites and Asiatic knotweed and Haloxyfop eradicated spartina, Manchurian wild rice and saltwater paspalum. Metsulfuron has eradicated sagittaria species, yellow flag and provided very good control of alligator weed (potentially imazapyr could give better control but yet to be trialled in aquatic situations).

5.15.6 Implementation - *Methods employed*

Treatment should follow herbicide label requirements.

5.15.7 Operation, maintenance, monitoring and reporting

There is requirement to fulfil EPA conditions, including reporting.

5.15.8 Practical considerations - *e.g. social issues, access constraints, consent requirements, health and safety concerns*

None of these herbicides are bio-accumulative. Evidence presented to the EPA on risks to aquatic and wetland flora and fauna and environmental fate of the four herbicides concluded no more than minor impacts were evident from studies to date.

5.15.9 Financial costs

Likely to be similar to other herbicides.

Cost estimate per ha*

(i) Start-up / implementation (once-off): Moderate
(ii) Operation & maintenance (annual): Low
(iii) Monitoring: Low
(iv) Decommissioning, if relevant (once-off): NA
(v) Overall cost over the lifetime of the intervention (maximum of 25 years): Moderate

* Costs do not include approvals, consent costs, public notifications, compliance monitoring, or reporting, which are likely to vary on a situational basis.

5.16 Natural herbicides

5.16.1 Description and overview

A range of ‘organic’ herbicides using plant oils are available that share similar characteristics. Here we focus on products which have the most available information.

Organic Interceptor™ (also marketed as BioWeed™ Control) is a certified organic herbicide for control of annual terrestrial weeds, grasses and for burn-off of perennial species. It is a non-selective, contact herbicide, containing 680 g per litre of emulsifiable pine essence which causes rapid plant desiccation (James et al 2002).

Organic Interceptor™ requires high levels of plant coverage for best results. When applied correctly it has a rapid knockdown effect and can achieve similar rates of weed control to glyphosate on a range of small terrestrial weeds (James et al 2002).

Because there is limited information relating to toxicity, fate and impacts in aquatic systems, use of organic herbicides should be avoided where there is a risk of water contamination.

5.16.2 Application - *In what situations can the option be applied?*

Organic herbicides may be used in terrestrial (e.g. dry banks) situations around stormwater devices where vegetation removal is sought (e.g. flood spillways), and where public objection to ‘chemical’ herbicide use is significant. The NZ Novachem Agrichemical Manual (2012) lists Organic Interceptor™ use in ‘waterways, drains, drain banks: apply before spring and autumn rains’.

Suggested target weeds include some marginal aquatic species (e.g. crack willow, arum lily), with control at the seedling stage emphasised and higher application rates recommended. Burn-off occurs most rapidly in warm, sunny weather whereas cold conditions slow the rate at which control is achieved.

5.16.3 Constraints - *In what situations can the option not be applied?*

Organic herbicides are not registered for control of submerged plants. During initial screening against three submerged weeds (NIWA unpublished data), Organic Interceptor™ required long contact times (≥10 hrs) and high concentrations (≥ 2,000 ppm) for total control. This concentration is several orders of magnitude higher than that required for the herbicide diquat (2 ppm) which is registered for use in aquatic systems (see 5.13; Diquat).

The Material Safety Data Sheet (for BioWeed™ Control) states ‘Expected to have no more than minor effects on aquatic organisms’. The half maximal effective concentration, EC_{50}, for fish was stated as 6.3 mg per L (96 hours exposure) and 10.9 mg per L for daphnia (after 48 hours exposure). This compares with equivalent EC_{50} for glyphosate of 180 mg per L for fish, and 930 mg per L for daphnia. Therefore, use of the organic product appears to have a greater risk for aquatic life than the use of glyphosate.

There is a rain-free period of three hours following application (commercial application manual, James and Rahman 2005). Silt-laden plants will reduce the emulsions effectiveness. Organic Interceptor™ ‘contains a natural, plant based surfactant’ (commercial application manual).

Significant regrowth of weeds is possible in as little as one to two weeks under agricultural applications, but potential for use in the urban situation was recognised (Johnson et al undated).

5.16.4 Requirements - *What other options / practices might be required in conjunction with this option?*

Mowing or slashing can be timed to reduce the weed canopy, promote young growth, increase spray coverage and improve efficacy. Low rate combinations of Organic Interceptor™ (10% of 680 g pine oil per l) and glyphosate (1% of 360 g per l) are suggested by the manufacturer for ‘synergistic effect’ at an application rate of 450 litres per hectare, although this defeats the ‘organic’ advantage.

5.16.5 Track record - *Where has this option been successful / unsuccessful?*

Organic Interceptor™ has provided adequate control for drains near organic farms in the Wairau Plains, Marlborough District.

5.16.6 Implementation - *Methods employed*

Use of herbicides should be always be guided by label information and/or manufacturer’s directions (see: Commercial application manual).

Organic herbicides can be applied using the same application equipment as other herbicides. Application methods include manual or powered sprayer with an emphasis on ensuring good mixing, high pressure delivery and coarse spray for maximal coverage and penetration.

However, applicators will be dealing with high volumes of product as higher rates of a.i. are required for organic herbicides than many other herbicides. For example, mixing rates for Organic Interceptor™ are stated as 10% to 20% product by volume. Effective application rates for Organic Interceptor™ were found to be 50–100 kg pine essence per ha on newly emerged weeds in freshly cultivated ground, and 150–450 kg per ha for established weeds (James et al 2002). This compares with application rates for glyphosate of 2.16-4.32 kg a.i. per ha.

Two applications, in spring and in autumn are recommended by the manufacturer with greater efficacy reported for young stages of plant growth.

5.16.7 Operation, maintenance, monitoring and reporting

Ongoing monitoring is required to identify the best timing of application, at the seedling stage, and confirm the level of control achieved.

5.16.8 Practical considerations *- e.g. social issues, access constraints, consent requirements, health and safety concerns*

Safety precautions are the same as for handling safety of any herbicide and include protective spray gear (goggles, respiratory mask, clothing and boots) while mixing and spraying. High volumes are required for application.

5.16.9 Financial costs

Product costs for Organic Interceptor™ are approximately $690 per ha. At an assumption of 2-3 applications per year and application costs of $100 per ha (costs variable, depending on application method) annual costs are likely to be $2,370 per annum.

Cost estimate per ha

(i) Start-up / implementation (once-off): Low
(ii) Operation & maintenance (annual): Low
(iii) Monitoring: Low
(iv) Decommissioning, if relevant (once-off): NA
(v) Overall cost over the lifetime of the intervention: Very high (25 year period)

Chemical control options for algae

5.17 Nutrient inactivation products

5.17.1 Description and overview

Nutrient inactivation products (also known as sorption media) can be used to remove nutrients from stormwater by surface bonding to the media or incorporation into the media. Here we focus on sorption media that may be applied directly to the water body to remove nutrients *in situ* (although sorption media may also be used in biofilters to remove nutrients from water prior to entering a pond). Nutrient inactivation products may be applied to ponds or lakes as an active capping agent (in contrast to a passive capping agent - see section on sediment lining). Active capping agents are designed to reduce sediment nutrient release. Active capping agents, such as aluminium sulphate (alum), modified zeolite (e.g. Aqual-P™), calcite, or clays such as bentonite that have been modified (e.g. Phoslock™), irreversibly bind dissolved inorganic phosphorus (and to some extent ammonium in the case of modified zeolite) to prevent release under anoxic conditions (Robb et al 2003; Berg et al 2004; Faithfull et al 2005; Drabkova 2007; Hickey and Gibbs 2009; Yuan et al 2009; Jančula and Maršálek 2011).

Figure 10 Modified zeolite (now known as Aqual-P) application in Lake Okaro, Bay of Plenty.

Figure 11 Aerial view of Aqual-P (modified zeolite) application in Lake Okaro, Bay of Plenty (Photo: Andy Bruere).

5.17.2 Application - *In what situations can the option be applied?*

Sediment capping with a nutrient inactivation product is likely to be most effective in deep ponds or lakes that are **anoxic** (in the bottom waters) for part or all of the year depending on dose rates.

5.17.3 Constraints - *In what situations can the option not be applied?*

In shallow systems that are affected by high rates of wind-driven resuspension, capping agents are likely to be quickly reworked into the bottom sediments and buried rendering the material largely ineffective. The toxicity of some of the nutrient inactivation products needs to be considered. For example, although research has indicated that there was little effect of modified zeolite on phytoplankton and zooplankton community composition, the study used a relatively low dose rate and coarse particle size; finer particles and a higher dose rate would likely affect filter-feeding zooplankton grazers (Özkundakci et al 2011). Adverse food web effects (e.g. reductions in grazers, fish kills) have been indicated in some Alum and Phoslock™ applications (Robb et al 2003; Paul et al 2008).

5.17.4 Requirements - *What other options or practices might be required in conjunction with this option?*

As the nutrient inactivation products will become buried with fresh sediment and organic matter from the catchment, it is important that this option is implemented in conjunction with attempts to reduce catchment sediment and nutrient loading.

5.17.5 Track record - *Where has this option been successful or unsuccessful?*

Active capping agents have been used in several Te Arawa / Rotorua lakes in an attempt to reduce internal nutrient loading (Özkundakci and Hamilton 2006; Paul et al 2008; Burns et al 2009; Özkundakci et al 2010). In Lake Okaro an application of modified zeolite was considered to be moderately effective in lowering internal phosphorus loading (Özkundakci et al 2010) (Figure 10, Figure 11). Suppression of phosphorus release from sediments has been observed in Lake Okareka following treatment with Phoslock™, although there was no decrease in water column phosphorus concentrations (Burns et al 2009).

5.17.6 Implementation - *Methods employed*

Active capping agents should be applied when the lake is fully mixed, when dissolved reactive phosphorus has been sequestered by sediments (i.e. before major releases occur into the water column). Alum is often applied as a solution to the lake surface water and the resulting floc then settles to the lake bed. As it binds with dissolved reactive phosphorus in the water column it also act as a flocculant (see 5.18; Flocculation). Because the pH of natural water also substantially influences the aquatic chemistry of aluminium it is important that jar tests are done to ensure adequate buffering to maintain pH above 6.5 (but below 8.5) to allow floc formation and to prevent formation of toxic Al^{3+} ions (Paul et al 2008). It should be noted that algal blooms can cause an elevation in pH outside of the required range, and may also cause further release of phosphorus and ammonium from sediments (Gao et al 2012). Furthermore, aluminium speciation in water is complex and not all the aluminium species formed may be available for nutrient uptake. Modified zeolite is a granular active capping agent that does not require buffering and recommended dosing is c. 350 g m^{-2} but is also dependent on the grain size of the material (Hickey and Gibbs 2009). Detailed descriptions of application methods are available in Hickey and Gibbs (2009).

5.17.7 Operation, maintenance, monitoring and reporting

Active capping agents will become buried by sediment and detritus within a few years of application, and this new sediment layer can release nutrients that can fuel algal growth (Hickey and Gibbs 2009). The rate of burial will be dependent on sediment input, thus new developments in the catchment of the stormwater system will likely reduce the efficacy of this sediment capping option and require new capping material to be applied. For this reason, it is generally accepted that application of capping agents to control internal nutrient loading in lakes should be carried out in conjunction with efforts to reduce external nutrient loading.

5.17.8 Practical considerations - *e.g. social issues, access constraints, consent requirements, health and safety concerns*

Application of active capping agents requires approval under the Hazardous Substances and New Organisms (HSNO) Act 1996, and consent under the Resource Management Act 1991 (RMA). For some systems approval may also be required from Department of Conservation. An **AEE**, including social and cultural values, is likely to be required.

5.17.9 Financial costs

Costs will be partially dependent on product used and approvals/consents required (for example modified zeolite is likely to be high cost, whereas calcite could be low cost).

(i) Start up: Low - High
(ii) Operation and maintenance: Low
(iii) Monitoring: Low
(iv) Decommissioning: NA
(v) Overall cost: Moderate - High

5.18 Flocculation

5.18.1 Description and overview

Flocculants can be used to cause rapid sedimentation of algal cells and/or phosphorus (Figure 12). Flocculation and sedimentation of harmful algal blooms (HABs) with clay has been investigated in several countries (Atkins et al 2001; Beaulieu et al 2005; Pan et al 2011b). The clay particles coagulate with algal cells and then the rapidly settling flocs further entrain algae, resulting in a floc layer that accumulates at the bottom of the pond or lake. The flocculation of algae is more effective at low flow speeds and consolidation of the floc layer over time can reduce resuspension (Beaulieu et al 2005). Chemical flocculants, such as polyaluminium chloride (PAC) or chitosan, can be added to the clays to further increase their effectiveness (Sengco and Anderson 2004; Pan et al 2011b).

Flocculants that bind with dissolved reactive phosphorus in the water column include products that may also act as active capping agents when they settle to the pond or lake bed, e.g. alum and Phoslock™ (see 5.17; Nutrient inactivation products).

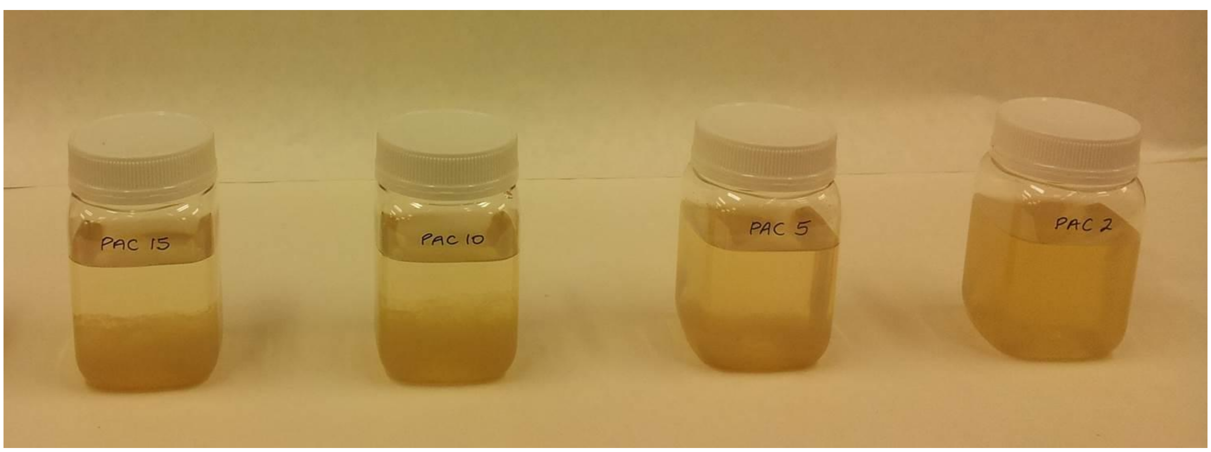


Figure 12 Testing efficacy of a flocculant (PAC; polyaluminium chloride) at different concentrations (2 – 15 mg L-1) for potential application in Lake Oranga (University of Waikato).

5.18.2 Application - *In what situations can the option be applied?*

Flocculants should be applied to the pond or lake when nutrient or algae concentrations in the water column are high. However, if using alum, consideration needs to be given as to the effect of algal blooms on the pH of the entire water column to ensure that it is within limits (c. 6.5 to 8.5) for effective flocculation.

5.18.3 Constraints - *In what situations can the option not be applied?*

Flocculation will result in the smothering of sediment (and associated fauna and flora) with algae and the flocculation/capping agent. In addition, the toxicity of some of the flocculant products needs to be considered. There are known to be acute and chronic toxic effects of alum on benthic invertebrates and fish (Lamb and Bailey 1981; Gensemer and Playle 1999) and when there has been inadequate buffering, fish kills have occurred due to the direct toxicity effects of the Al^{3+} ion and/or low pH. Effects studies for various NZ freshwater species have been carried out for Phoslock™ and have been reviewed in Hickey and Gibbs (2009). A risk assessment will likely be required on a case-by-case basis to determine the ecological risks for the whole system.

5.18.4 Requirements - *What other options or practices might be required in conjunction with this option?*

As for active capping agents, application of flocculants is likely to be a short- to medium- term fix for control of algae growth. Thus, it is important that this option is implemented in conjunction with attempts to reduce catchment sediment and nutrient loading.

5.18.5 Track record - *Where has this option been successful or unsuccessful?*

Alum flocculation of two stream inflows to Lake Rotorua is currently used in an attempt to reduce phosphorus loading (Burns et al 2009). Alum has been used in Lake Okaro with no clear evidence of marked improvement in water quality (A. Bruere Bay of Plenty Regional Council pers. comm.); one alum treatment appeared to be associated with a pulse of ammonium and reduced pH, and an algal bloom was also observed post treatment (Paul et al 2008). Modified (with chitosan) local soils were successfully used to flocculate a HAB in a bay of Lake Taihu (China) as part of an attempt to reduce the effects of eutrophication and restore submerged macrophytes (Pan et al 2011a).

5.18.6 Implementation - *Methods employed*

As for “Nutrient Inactivation Products.”

5.18.7 Operation, maintenance, monitoring and reporting

Flocculants are typically used in one-off applications, so there may be no maintenance requirements. If potentially toxic flocculants are to be used in frequent or repeated applications, consideration should be given to the ecological effects of cumulative dosing. Dosing of stream inflows to Lake Rotorua represents a potentially more benign method of avoiding or mitigating potential toxicity or pH issues associated with one-off applications of alum but will require monitoring to test for long-term issues (e.g. long-term buildup of aluminium in the bottom sediments).

5.18.8 Practical considerations - *e.g. social issues, access constraints, consent requirements, health and safety concerns*

As for active capping agents, application of flocculants typically requires approval under the Hazardous Substances and New Organisms (HSNO) Act 1996, and consent under the Resource Management Act 1991 (RMA). For some systems approval may also be required from Department of Conservation. An **AEE**, including social and cultural values, is likely to be required.

5.18.9 Financial costs

Costs will likely be highly variable (dependent on for example the product used and approvals/consents required).

(i) Start up: Moderate - High
(ii) Operation and maintenance: Low
(iii) Monitoring: Low
(iv) Decommissioning: NA
(v) Overall cost: Moderate - High

Physical control

Physical control options for both plants and algae

5.19 Physical shading

5.19.1 Description and overview

Increased shading may control water weeds and nuisance algae by limiting the amount of light available to aquatic plants and algae for growth. Here we will focus on shading achieved through the use of material covers (e.g. black plastic, floating islands), and riparian vegetation (trees).

Surface covers deployed over the water surface are often constructed using the same materials as bottom liners (Figure 13) and to be effective need to filter out approximately 90% of ambient light (Authors' observations). They can be used to exclude light from drainage systems while enabling water flow underneath, and in larger ponds floating 'rafts' have the advantage of being able to be moved around. Surface covers would be best suited for the spot treatment of plants and/ or algae in small areas. Shading lasts only as long as covers are in place.

Figure 13 Surface cover fitted over a ditch system to control aquatic vegetation (Photo: R. Wells, NIWA).

Floating islands typically consist of a hydroponic mat-like platform (Figure 14), with plants growing on the surface and roots hanging in the water. The primary use for floating islands is water quality treatment of storm and other waste waters (Headley and Tanner 2012). While floating islands would be expected to provide the same shading benefits as floating surface covers for the control of aquatic plants and algae, they are considered here only as an additional benefit to their primary purpose of water quality treatment.

Increased shading from trees and other emergent and floating vegetation can also exert a degree of control on some aquatic plants and algal growth by reducing light reaching the water column (Yeh et al 2011). For example, shading by riparian vegetation has been shown to be an important factor governing periphyton growth in streams (Mosisch et al 2001). Trees and other riparian vegetation also have additional benefits such as reducing water temperatures in wet ponds and mitigating concerns of thermal pollution in receiving waters (Auckland Regional Council 2003). Trees will require time to grow before affording shade, but it is a long-term option. To be most effective the vegetation needs to be on the north side of the water body and densely overhang the water body. Often weed growth still occurs though will not be as dense.

Figure 14 Floating island in place on motorway stormwater treatment pond in Silverdale, North Auckland. (Photo: C. Tanner, NIWA).

5.19.2 Application - *In what situations can the option be applied?*

Physical shading techniques are most suitable for narrow channels and smaller sized systems.

Care should also be taken to avoid growing trees close to dams or other structures where tree roots may impede on the structural integrity of a system.

5.19.3 Constraints - *In what situations can the option not be applied?*

The aerial extent of physical shading required for aquatic plant control is not feasible in large systems (e.g. >0.1ha or > 20 m wide). Storm flows and strong winds can dislodge shade covers and surface covers may interfere with recreational activities and be aesthetically unpleasing to look at. There may be issues with the safety of the public if covers are accessible (e.g. mistaken for a buoyant structure).

In situations where channels need to be kept clear to maintain water flows, trees placed close to waterways can input debris and branches if they are not maintained. Waterway access will also be impeded by the presence of tall growing vegetation. This can be an issue where machine access is required.

5.19.4 Requirements - *What other options / practices might be required in conjunction with this option?*

This option is stand alone depending on the degree of shading achieved.

5.19.5 Track record - *Where has this option been successful / unsuccessful?*

Weed matting suspended above the water has been used to control lagarosiphon in a Takaka stream by the Tasman District council and in a stream near Blenheim by the Marlborough District Council. This form of control was reported to be effective at the time but focus is now on the planting of trees and riparian vegetation to increase shade.

The use of floating vegetated islands is still limited in NZ with floating islands being trialled in a stormwater system in Rosedale, Bayside reserve, and Silverdale, Auckland region (Borne and Fassman 2011). In addition, a trial site in Lake Kaituna (Waikato) is being used to assess the potential for floating wetlands to reduce nutrient concentrations in detention ponds on agricultural land. Elsewhere, floating islands have been used for treatment of wastewater and in eutrophic lakes (e.g. Lake Rotoehu and Lake Rotorua, Bay of Plenty) to remove nutrients from the water column and suppress algal growth (Chang et al 2013; Stefani et al 2011; Zhu et al 2011).

5.19.6 Implementation *- Methods employed*

Surface covers may be constructed from polyethylene, PVC, polypropylene, nylon, synthetic rubber materials or fibreglass screens. Covers attached to floats could be moved about within small ponds or areas. For the best results, shading techniques should be established in early spring, at the start of the growth season for water weeds and nuisance algae.

The selection of trees and other riparian vegetation for shading purposes will largely depend on the area and density of shading required and current information for this purpose is limited. It is recommended that a planting plan be developed for an area with guidance from landscape or planting professionals to establish the best species and place to plant for maximum shading affect (e.g. northern side of a waterbody).

5.19.7 Operation, maintenance, monitoring and reporting

The use of surface covers would require most effort at the installation phase. Following this, regular checks would need to be made to ensure floating covers remain in place and un-damaged. Additional effort would also be required in the event that a cover needs to be removed or re-instated.

Trees and other riparian vegetation will require a degree of on-going maintenance to ensure establishment and survival.

5.19.8 Practical considerations *- e.g. social issues, access constraints, consent requirements, health and safety concerns*

Surface covers can be difficult to apply over large areas or over obstructions and can be aesthetically unpleasing. Some materials may also be degraded with sunlight.

Time scales for growth of new trees will differ significantly; the planting of an exotic tree (E.g. pines, eucalyptus) may require 10 or more years to shade a small area or system while slow-growing native trees may take 20-25 years.

5.19.9 Financial costs

Surface covers would be similar cost to bottom lining at $30,000 per ha. Riparian planting with native plants (planted PB2 grade, including $8,000 maintenance for 3 years) is estimated at $20,500 per ha (de Winton et al 2010).

Cost estimate per ha

(i) Start-up / implementation (once-off): Moderate

(ii) Operation & maintenance (annual): Low to moderate

(iii) Monitoring: Low

(iv) Decommissioning, if relevant (once-off): Low

(v) Overall cost over the lifetime of the intervention (maximum of 25 years): Moderate to high

5.20 Shading by dyes

5.20.1 Description and overview

Aquatic dyes or pond colorants are usually non-toxic vegetable dyes with blue and black colours available. The most common use for aquatic dyes is for aesthetic reasons in ornamental and golf course ponds.

Dyes might inhibit submerged plants and algae growth in ponds and small lakes by limiting light penetration into the water (shading) and by absorbing wave lengths within the photosynthetically active spectrum of light. Black colours are regarded as most beneficial for weed and algal control. Dyes must persist for several months in a system to be effective and are likely to need re-applying at regular intervals.

5.20.2 Application - *In what situations can the option be applied?*

Use of dyes will only be feasible in low volume systems with little water exchange.

5.20.3 Constraints - *In what situations can the option not be applied?*

The duration for treatment is a function of water retention time, so dyes are not recommended for use in on-line systems or systems which receive large volumes of water when it rains.

Shallow and emergent plants or buoyant blooms of phytoplankton will not be affected by dyes. The effect of shading on particular problematic plants and algae is unclear since research is very limited.

5.20.4 Requirements - *What other options / practices might be required in conjunction with this option?*

Dyes are best used as a control option in conjunction with other pond management methods (e.g. to retard growth after harvesting, drawdown etc.).

5.20.5 Track record - *Where has this option been successful / unsuccessful?*

We have not seen effective weed control using dyes. There is little information on the efficacy of dyes for aquatic plant and algal control in NZ although they have been used in ornamental ponds and golf course ponds for many years.

Only one published study into the efficacy of shading by dyes as an algae and vegetation control option. The dye Aquashade® (applied at twice the recommended dose) was not found to significantly affect algal or macrophyte growth in small, shallow fish culture ponds in Arkansas, USA (Ludwig et al 2010).

5.20.6 Implementation - *Methods employed*

Dyes are applied as a whole of system treatment. A range of products are advertised (e.g. Aquashade®, DyoFix) and use should be guided by information on the product label. For the best results dyes should be applied in early spring, at the start of the growing season for water weeds and nuisance algae.

Adding any product to water will require permission from the Regional Council.

5.20.7 Operation, maintenance, monitoring and reporting

As strong sunlight and rainfall both affect dye concentrations in a waterbody there is no definitive answer as to how long dye will last. The suppliers of DyoFix suggest a visual assessment of the need for retreatment, by comparing 'depth of colour' in water sampled at monthly intervals with stored samples taken immediately after treatment (and stored in the dark). Retreatment is recommended when 'depth of colour' is about half the strength of the initial sample.

5.20.8 Practical considerations - *e.g. social issues, access constraints, consent requirements, health and safety concerns*

Most aquatic dyes are reported to be non-toxic to humans and to most aquatic organisms including invertebrates and fish. Aquashade® is registered for aquatic use with the United States Environmental Protection Agency (EPA 2005).

5.20.9 Financial costs

Based on dye costs, estimated $1,125 per ha, assuming 3 treatments per annum, is $3375.

Cost estimate per ha

(i) Start-up / implementation (once-off): Low

(ii) Operation & maintenance (annual): Low

(iii) Monitoring: Low

(iv) Decommissioning, if relevant (once-off): NA

(v) Overall cost over the lifetime of the intervention (maximum of 25 years): Very high

5.21 Manual harvesting

5.21.1 Description and overview

Manual harvesting techniques in aquatic systems include hand weeding, raking and netting.

Hand weeding is the selective removal of individual plants, including their root systems, and can be used effectively for removing limited new weed incursions, for small, localised weed infestations (Bellaud 2009), and for removing undesirable species in mixed plant communities. It is not practical once infestations expand, and it is very labour intensive. Hand weeding may be possible by wading in shallow systems, but scuba or snorkel diving would be required in water depths of 1.2 m or more (Bellaud 2009).

Raking and netting reduce the biomass of submerged weeds or filamentous algae within limited areas. They may be carried out by throwing and dragging equipment from the bankside in small systems, or otherwise deployed from a boat. The period of control is likely to be similar to mechanical harvesting, repeated more than once in a growth season to prevent surface growth.

5.21.2 Application - *In what situations can the option be applied?*

Hand weeding is appropriate for weed eradication in situations where a target weed can be easily identified (e.g. sufficient water clarity) and is distributed at a low density of <125 shoots per 0.1 ha (Bellaud 2009), or where patches do not exceed 1 m^2.

Raking and netting can be applied in small systems (>0.1 ha) or limited areas of large systems.

5.21.3 Constraints - *In what situations can the option not be applied?*

Hand-weeding ceases to be a viable option for eradication once weed colonisation becomes too advanced. Water clarity and impacts from sediment re-suspension may limit its application.

Raking and netting should not be used where weed spread may be advanced by disturbance and fragmentation.

5.21.4 Requirements - *What other options / practices might be required in conjunction with this option?*

Hand weeding may follow suction dredging or bottom lining efforts to 'mop up' plants established from fragments after a weed eradication attempt.

5.21.5 Track record - *Where has this option been successful / unsuccessful?*

Hand weeding has been used to remove weed infestations in the USA (Bellaud 2009). In Lake Wanaka and Lake Waikaremoana, hand weeding by scuba divers has removed isolated plants of lagarosiphon, or followed up after suction dredging during site specific eradication attempts.

Raking and netting has been used in small ornamental lakes.

5.21.6 Implementation - *Methods employed*

Hand weeding for eradication should be carried out as soon as possible after the detection of a new weed incursion. Only scuba divers certified by the Department of Labour should be used. Care in completely removing plants (e.g. avoiding shoot breakage, incorporating root crown) is very important. Demarcation of an underwater search grid (i.e. lines and marker buoys) will make detection more effective. Raking and netting can be undertaken as needed.

5.21.7 Operation, maintenance, monitoring and reporting

A weed eradication attempt will require longer-term surveillance to confirm the last fragment has been successfully removed. Typically surveillance in aquatic systems may be required for 3-5 years after the last removed fragment before eradication is confirmed.

5.21.8 Practical considerations - *e.g. social issues, access constraints, consent requirements, health and safety concerns*

Safety of personnel operating around and on aquatic systems is paramount (e.g. personal floatation devices, boat handling skills).

5.21.9 Financial costs

Costs estimated for two hand weeding treatments to achieve weed eradication would be $20,000 per ha. Costs for use of manual harvesting for on-going control are likely to be prohibitively expensive unless community groups can be involved that are willing and able to undertake weed clearance (e.g. marginal or shallow water plants).

Cost estimate per ha

(i) Start-up / implementation (once-off): Low

(ii) Operation & maintenance (annual): Low

(iii) Monitoring: Low

(iv) Decommissioning, if relevant (once-off): NA

(v) Overall cost over the lifetime of the intervention (maximum of 25 years): Moderate

Physical control options for plants

5.22 Mechanical harvesting

5.22.1 Description and overview

The use of machinery to cut, collect, and transport submerged weed biomass for on-shore disposal is considered here. Other weed cutting or rototilling options that do not involve the removal of weed biomass will be unsuitable for stormwater systems. This is because there is no passive means of weed disposal such as occurs in natural systems, where cut weed may be removed by strong currents, or lost to extensive areas deeper than suitable for plant survival and growth. Therefore, unless actively removed, the cut weed fragments are likely to re-establish and may even spread more widely. Mechanical treatments for emergent plants are given in 5.21; Manual harvesting and 5.23; Mechanical excavation.

Mechanical harvesters for submerged weeds (and large floating weeds) comprise watercraft fitted with cutter bars, conveyer belts to collect floating fragments and a temporary storage area (Figure 15). Harvesting is usually possible to a maximum of 2.0 m water depth (Wells et al 2000). Because re-growth can occur rapidly from residual rooted material, harvesting may need to be repeated two or three times in a growing season to prevent weed beds from occupying the entire water column and developing a canopy at the water surface (Wells et al 2000).

Figure 15 Mechanical harvester (Photo: John Clayton, NIWA).

Draglining involves towing a chain/cable between two tractors on either side of a system. It is less effective at removal of weed fragments than a harvester. However, it may have a longer effect due to the partial removal of root systems, but is more disruptive to aquatic ecosystems.

A mechanical digger can be used in small systems, where a modified or 'clam shell' bucket is used to remove submerged weed material, but sediment can be left relatively undisturbed.

All methods require the disposal of material removed from the system, with the range in costs largely depending on the difficulty and distance required for transportation of spoil.

5.22.2 Application - *In what situations can the option be applied?*

Harvesters would only be cost effective in larger systems with extensive submerged weed beds, and requires access for machinery as well as a suitable weed off-loading site (Haller 2009). System depth is not critical, with harvesters commonly able to operate in as little as 0.3 to 0.45 m water depth (Haller 2009).

Draglining requires machinery access to both sides of a waterway (Wells and Clayton 2005) and would be suitable for very narrow systems (c. <5 m width). Suitability for digger control also depends on the size/configuration of the system and reach of the digger arm.

5.22.3 Constraints - *In what situations can the option not be applied?*

Harvesters and draglining are unsuited to systems with uneven bottom contours or obstacles (Wells et al 2000). These options are not appropriate for early stages of weed invasion as it may speed fragment spread.

5.22.4 Requirements - *What other options / practices might be required in conjunction with this option?*

Harvesting is usually a stand-alone option.

5.22.5 Track record - *Where has this option been successful / unsuccessful?*

Numerous types of harvesting machines are widely used in the USA and China (Zhang et al 2008; Haller 2009). In NZ, imported harvesters have cut and removed nuisance weed growths twice a year from the slow-flowing Avon River, Christchurch (Wells and Clayton 2005). Elsewhere, a variety of locally designed cutting machines are operational, mostly purpose-built for use in small waterbodies, canals and drainage systems (Wells and Clayton 2005). Assessments of harvesting in the Waikato hydrolakes suggested 1 to 2 treatments per year would result in a reduced weed bed height (Howard-Williams et al 1988-1991; Howard-Williams et al 1996).

Draglining and diggers are commonly used in drainage systems (Hudson and Harding 2004).

5.22.6 Implementation - *Methods employed*

Harvesting operations are likely to require delineating/marking of the area for treatment and a briefing provided to operators.

Transfer of equipment between systems requires machinery hygiene protocols to be implemented to prevent the accidental spread of weed fragments.

5.22.7 Operation, maintenance, monitoring and reporting

Pre- and post-treatment inspections will be required to judge effectiveness, while monitoring will identify the timing for re-treatment, depending upon factors determining the speed of plant growth.

5.22.8 Practical considerations - *e.g. social issues, access constraints, consent requirements, health and safety concerns*

Harvesters are specialist pieces of equipment that come in different configurations for varied situations (Zhang et al 2008; Haller 2009). Access to an appropriate model for stormwater systems may be restricted or expensive in a small market like NZ.

Harvester, dragline and digger operations may remove large amounts of benthic fauna and fish associated with the weed, and produce high turbidity or anoxia (Wells and Clayton 2005; Haller 2009).

Harvesting submerged weed material may have benefits in terms of nutrient removal from systems (Section 6.2).

5.22.9 Financial costs

Costs of a harvester are approximately $2,000-4,000 per hectare (Wells et al 2000). Costs would vary considerably with the distance to a dump site, with up to half of costs related to disposal (Haller 2009).

Costs for draglining vary from $250 to $500 per hectare depending on whether a wheeled vehicle or tracked vehicle is required and the reach required from the edge of the waterway (Wells and Clayton 2005).

Costs for mechanical digger removal of submerged weeds start from about $1,000 per hectare (Wells and Clayton 2005).

Assuming 3 treatments per year, mechanical harvesting costs would be $3,000 to $12,000 per ha, per annum.

Cost estimate per ha

(i) Start-up / implementation (once-off): NA

(ii) Operation & maintenance (annual): Low to Moderate

(iii) Monitoring: Low

(iv) Decommissioning, if relevant (once-off): NA

(v) Overall cost over the lifetime of the intervention (maximum of 25 years): Very High

5.23 Mechanical excavation

5.23.1 Description and overview

A mechanical digger Figure 16) can be used in waterways to remove beds of emergent weeds (for submerged weeds see 5.22; Mechanical harvesting). Diggers may be particularly effective in reducing rhizomatous weed species that are difficult to control by other means. However, careful disposal of excavated material will be required. Eradication may be achieved by one operation, but it is likely that follow-up work (e.g. spot-spraying, hand weeding) will be needed.

Figure 16 Digger removal of yellow flag from a drainage system (Photo: R. Wells, NIWA).

5.23.2 Application - *In what situations can the option be applied?*

Eradication of new, spatially limited infestations of emergent weeds would be suitable by excavation. Access is required for digger equipment, and for the removal of spoil.

5.23.3 Constraints - *In what situations can the option not be applied?*

Use for control of extensive areas of emergent weeds is not cost effective. The design of some stormwater systems (e.g. lined ponds, structural embankments) will not be suitable for excavation. For instance, diggers can over-deepen and over-widen drains and often affect the sides of drains causing bank instability and loss of marginal habitat (Wells and Clayton 2005).

5.23.4 Requirements - *What other options / practices might be required in conjunction with this option?*

Herbicides appropriate for use on the target species may be used to remove remaining plants (see Chemical control).

Diggers have been responsible for the spread of invasive weeds to new sites, so machine hygiene is important (6.1; Machinery and materials hygiene), as is the careful disposal of spoil.

5.23.5 Track record - *Where has this option been successful / unsuccessful?*

Mechanical diggers are routinely used in drainage systems to remove marginal and submerged weeds as well as sediment, in order to maintain drainage flows.

5.23.6 Implementation - *Methods employed*

Operations are likely to require a delimitation of the weed species and possible marking of treatment areas. Digger operators will need to be briefed on the target site and plant, required depth of excavation (e.g. for removal of rhizome material) and hygiene considerations.

5.23.7 Operation, maintenance, monitoring and reporting

Follow-up for eradication attempts will involve regular inspection, and possible spot-spraying or hand weeding depending on the species and effectiveness of excavation.

5.23.8 Practical considerations - *e.g. social issues, access constraints, consent requirements, health and safety concerns*

This method often produces high turbidity and sometimes anoxia in adjacent waters (Wells and Clayton 2005). Contact regional councils before excavating in and around natural systems.

5.23.9 Financial costs

Costs are about $2,500 per ha (Wells and Clayton 2005). Assuming 2 treatments to eradicate a weed, cost is $5000 over lifetime.

Cost estimate per ha

(i) Start-up / implementation (once-off): Low

(ii) Operation & maintenance (annual): Low

(iii) Monitoring: Low

(iv) Decommissioning, if relevant (once-off): Low

(v) Overall cost over the lifetime of the intervention (maximum of 25 years): Low

5.24 Mowing

5.24.1 Description and overview

Mowing involves mechanical cutting of bank-side, or near-shore emergent plants using mowers, saw blades or 'weed eaters', and results in a reduction of plant biomass and height to a near ground level. For physical cutting of submerged weeds see 5.22; Mechanical harvesting.

Suppression of tall-growing bank-side weeds such as yellow flag, reed grass Californian bulrush, giant reed or spartina may be provided by on-going mowing, although extra effort may be required to cut these back initially. Repeated mowing over time may also deplete the reserves in underground **rhizomes** of some of these weeds and result in a lower stature weed beds (Derr 2008) and slower spread (Gusewell 2003). Maintenance mowing of drainage systems in NZ is usually undertaken 2-4 times per year (Hudson and Harding 2004).

5.24.2 Application - *In what situations can the option be applied?*

Mowing is suitable for situations where machinery access is allowed by the terrain. Mowing provides on-going maintenance control. Mowing may an appropriate alternative option in some situations where there is public concern over herbicide use (Derr 2008).

5.24.3 Constraints - *In what situations can the option not be applied?*

Eradication of marginal weeds is unlikely to be achieved by mowing on its own (Gusewell 2003; Derr 2008) and it is generally considered less effective than herbicides against rhizomatous weeds, such as phragmites. Although multiple mowing may act to reduce weed species, it is also likely to reduce the presence of other plants.

5.24.4 Requirements - *What other options / practices might be required in conjunction with this option?*

Mowing may be carried out prior to, or after herbicide application, although timing needs to be considered. Reduction of plant biomass by mowing prior to herbicide application may increase efficacy (Monteiro 1999), promote young growth more amenable to herbicide translocation (e.g. glyphosate isopropylamine) and reduce the environmental load of herbicide required.

5.24.5 Track record - *Where has this option been successful / unsuccessful?*

Mowing is widely used for drainage systems in NZ to maintain grass cover and for aesthetic purposes (Hudson and Harding 2004). It was been trialled for use in management of Manchurian wild rice in Northland, to reduce biomass before using herbicide, and was used to control yellow flag around Lake Rotoroa in Hamilton and phragmites in Napier.

5.24.6 Implementation - *Methods employed*

Mowing is widely used for the maintenance of grounds in parks and verges in Auckland, and requires no specialist equipment for application around stormwater systems, although safety around steep banks and inundated areas should be paramount. It may be carried as needed. The timing of mowing to deplete rhizome reserves may be more critical (Asaeda et al 2006), with most investigators suggesting late summer mowing and again in early spring.

5.24.7 Operation, maintenance, monitoring and reporting

Mowing is likely to be on-going at a similar cost each year.

Additional effort would be required for machine hygiene (6.1; Machinery and materials hygiene) at sites with weeds that are easily spread by fragments (e.g. alligator weed), seeds (e.g. yellow flag) or rhizome sections (spartina). Likewise if spoil is removed it must be disposed of carefully.

5.24.8 Practical considerations *- e.g. social issues, access constraints, consent requirements, health and safety concerns*

Safe access for machinery and staff should be paramount, and wet or steeply sloped areas should be avoided. Mowing with a saw blade poses a particular hazard to the user (Derr 2008).

Financial costs

Costing based on $175 per ha, with 2 to 4 maintenance mowings per year is $700 per ha per annum.

Cost estimate per ha

(i) Start-up / implementation (once-off): NA
(ii) Operation & maintenance (annual): Low
(iii) Monitoring: Low
(iv) Decommissioning, if relevant (once-off): NA

(v) Overall cost over the lifetime of the intervention (maximum of 25 years): Moderate

5.25 Bottom lining

5.25.1 Description and overview

The bottom lining of systems using 'benthic barriers' excludes light for submerged plant growth and removes access to substrates for rooting. This option is suitable for a one-off weed eradication, or to provide medium-term control (years) in reducing vegetation biomass or maintaining open water habitat. The outcome depends upon the extent of installation and the properties of the material used.

Varied materials are used for benthic barriers, each having advantages and disadvantages (Caffrey et al 2010). The most common is plastic or polyethylene sheeting (Helfrich et al 2009, Caffrey et al 2010). Woven weed cloth (Figure 17), jute hessian, and fibreglass panels have also been used. Linings may be covered by a layer of mineral soils (sand, gravel, clay), or in some situations these substrates have been used on their own, but have proven less effective (Helfrich et al 2009).

Figure 17 Placement of weed cloth held in place by sand bags (Photo: R. Wells, NIWA).

5.25.2 Application - *In what situations can the option be applied?*

Benthic barriers are appropriate for submerged weed beds in small areas or systems, with the upper suggested feasible size for installation being 0.4 ha (U.S. Army Corps of Engineers 2012).

5.25.3 Constraints *- In what situations can the option not be applied?*

Benthic barriers are not effective against free-floating plants such as *Azolla* species, and are less effective for the non-rooted plant, hornwort, than other weeds.

Steep slopes or areas with numerous obstacles are difficult to bottom line. Regrowth of some vigorous emergent plants can penetrate through benthic barriers. On organic sediments, the build-up of gas beneath the lining can dislodge it unless the liner is perforated.

The influence of benthic barriers on sediment sources of nutrients for algae needs to be considered. Effects of benthic lining on sediment-water nutrient dynamics are related to the permeability of the material used and sediment type, but might include increased or decreased release of reduced elements from beneath the barriers (Eakin and Barko 1995). Decreased dissolved oxygen and increased NH_4 beneath the lining may reduce macroinvertebrate densities (Ussery et al 1997).

5.25.4 Requirements - *What other options / practices might be required in conjunction with this option?*

Temporary dropping of water levels may be advantageous for the installation of benthic linings. Likewise the removal of weed biomass by harvester or herbicide might be required prior to laying benthic barriers.

5.25.5 Track record - *Where has this option been successful / unsuccessful?*

Benthic barriers, usually plastic sheeting or synthetic fabric, has been used widely in the US, where it has been proved effective against milfoil (*Myriophyllum spicatum*) infestations (Laitala et al 2012). Biodegradable jute matting was successfully used against infestations of lagarosiphon in an Irish lake (Caffrey et al 2010).

In NZ, weed matting or polythene sheeting was used to reduce the extent of a lagarosiphon infestation in Rosie Bay, Lake Waikaremoana, a water lily in Lake Okareka and at Raupara water gardens, Coromandel, and hydrilla (*Hydrilla verticillata*) in localised areas of Hawkes Bay lakes. Weed matting or polythene has been used for **amenity control** of submerged weeds at boat ramps and jetties.

In the Auckland region bottom lining is reported to have been used in a pond on Great Barrier Island to control egeria (AC records).

5.25.6 Implementation - *Methods employed*

Here, we consider two permeable materials; polypropylene woven weed matting, and 'hessian' or coconut fibre matting. These mesh materials are easier to install, allow dissolved gases to escape and may enable macroinvertebrate species to migrate between the sediments and water column (Caffrey et al 2010).

Polypropylene woven weed matting is durable for situations where long term control of vegetation is required (e.g. at screens and outlets). Synthetic geotextile fabric panels were successful in removing invasive milfoil after eight weeks placement in a US lake (Laitala et al 2012). Note, however, that in systems receiving high suspended sediment loads, the lifespan of control will be reduced by the build-up of sediment over linings. One study of plant re-colonisation performance suggests benthic barrier maintenance should include sediment removal when sediment reaches a depth of 4 cm (Laitala et al 2012). Non-biodegradable materials may ultimately need to be removed, which might then allow submerged plants to rapidly recolonize (Eichler et al 1995).

Hessian or coconut fibre matting is a recently tested material. Jute hessian was successful in controlling lagarosiphon within as little as four months at lake sites (Caffrey et al 2010). NIWA trials showed a denser hessian material and coconut fibre could successful remove lagarosiphon, egeria and hornwort within five months (Hofstra et a al. 2010). These biodegradable materials would be advantageous in situations where eradication of invasive species is sought in small systems by temporary bottom lining. Submerged use suggests jute hessian lasting c. 7-10 months before disintegration (Caffrey et al 2010). An additional advantage is the weave of these materials allows native plants to grow through from vegetative or seed sources (Caffrey et al 2010; Hofstra et al 2010). Benthic barriers may provide some benefits analogous to geotextiles for bank and sediment stability, and encourage native plant colonisation (Caffrey et al 2010).

Installation of benthic barriers may require snorkel or scuba divers to assist in placement (e.g. to ensure overlapping strips) particularly in deeper systems (e.g. >1.8 m). Linings will also need to be weighed down by sand bags, rocks, or else pined in place. This is important to ensure linings will not shift or block outlets. Alternatively, a layer of sand, or gravel, that provides less suitable rooting media for plants, may hold linings in place.

Installation may be best timed for when plants have lower stature in late winter to early spring.

5.25.7 Operation, maintenance, monitoring and reporting

Most effort is involved at the installation phase. Annual checks will ensure the lining has not shifted or if any repairs are required (depending on required length of control). Gas build up beneath linings (reduced in the case of woven materials) may necessitate venting and re-anchorage. Removal of sediments may need to be carried out more frequently than usually scheduled in stormwater systems. Additional effort would be required in the event that linings need to be removed.

5.25.8 Practical considerations *- e.g. social issues, access constraints, consent requirements, health and safety concerns*

Use in natural systems will need to be checked with regional councils.

5.25.9 Financial costs

Cost estimate of \$3 m^2 and assuming 1 treatment is \$30,000 per ha, excluding any removal of sediments.

Cost estimate per ha

(i) Start-up / implementation (once-off): High
(ii) Operation & maintenance (annual): Low
(iii) Monitoring: Low
(iv) Decommissioning, if relevant (once-off): Low
(v) Overall cost over the lifetime of the intervention (maximum of 25 years): High

5.26 Suction dredging

5.26.1 Description and overview

Suction dredging removes submerged weeds, including their root systems (Figure 18), with some level of control over the depth of sediment removed. Suction dredging can effectively reduce the biomass of rooted weeds, and may rehabilitate sediments to make them less suitable for supporting large weed beds. This can give effective control for up to three years in lagarosiphon beds in lakes (Wells et al 2000). However, it is unlikely to achieve weed eradication alone, because of the requirement to remove all weed fragments. Suction dredging, combined with follow-up hand weeding, has eradicated weeds from some sites in lakes.

Figure 18 Diver directed suction dredging in Lake Wanaka (Photo: J. Clayton, NIWA).

5.26.2 Application - *In what situations can the option be applied?*

Suction dredging is useful for removal of targeted infestations from areas at an early stage of establishment (i.e. new weed incursions that are too big to hand-weed). It may have additional benefits where sediment removal is an advantage, such as part of long term maintenance of detention basins or forebays. It is suitable for small systems (<0.1 ha) or partial areas of larger systems.

Vehicular access is required to the system for equipment. Shore-based suction dredging may be possible in small systems (<0.1 ha), but larger systems are likely to require access for a barge.

5.26.3 Constraints - *In what situations can the option not be applied?*

The high relative cost and slow removal rate makes this option unsuitable for general weed control. Suction dredging is not effective against non-rooted hornwort with re-establishment possible in as short as two months (Wells et al 2000).

Temporary impacts on water quality are likely and the impact on sediment-based biota depends on the degree of disturbance. Operations may be hampered by poor water clarity. Suction dredging also cannot be easily used in hard-bottomed or rocky substrates. Disposal of spoil can be an issue, particularly if high concentrations of heavy metals are present in sediments. Disposal may be cheaper if it can be done on site (e.g. behind a silt curtain).

5.26.4 Track record - *Where has this option been successful / unsuccessful?*

Suction dredging was used to eradicate a submerged weed from a 610 m length of river in Texas, USA (Alexander et al 2008). In Lake Wanaka, NZ, suction dredging has been used since 1980 to remove outlier colonies of lagarosiphon (Wells and Clayton 2005) and for public amenity areas like boat ramps and jetties, to minimize the risk of transfer within the lake and to nearby uninfected water bodies. Suction dredging has also been used for amenity purposes in the Rotorua Lakes. More recently, suction dredging has been used in Lake Waikaremoana to remove new incursions of lagarosiphon.

5.26.5 Implementation - *Methods employed*

Areas of dense weed growth need to be delineated for removal. A diver-directed suction dredge, using a venturi suction pump, uproots the aquatic weeds and discharges them into a receptacle such as a floating barge or fine mesh bags for later disposal. A clearance rate of up to 20 days per ha is likely in dense weed beds. Disposal may be on-site if an appropriate area exists, else there will be additional transportation costs. Careful disposal of aquatic weeds is required to avoid unintentional spread.

5.26.6 Operation, maintenance, monitoring and reporting

Operations are likely to be one-off, particularly if the goal is weed eradication (with follow-up by hand weeding) or sediment removal.

No maintenance is required, although monitoring of the effectiveness and duration of control is recommended on a seasonal to annual basis.

5.26.7 Practical considerations - *e.g. social issues, access constraints, consent requirements, health and safety concerns*

Dredging operations in natural waterbodies are likely to require resource consent.

Harvesting submerged weed material may have benefits in terms of nutrient removal from systems (Section 6.2).

5.26.8 Financial costs

Costs are about $7 - 10,000 per ha in Lake Wanaka and $15-20,000 per ha Rotorua lakes (Wells et al 2000). Assuming a one off treatment, estimated cost is $20,000 over lifetime.

Cost estimate per ha

(i) Start-up / implementation (once-off): Moderate
(ii) Operation & maintenance (annual): NA
(iii) Monitoring: Low
(iv) Decommissioning, if relevant (once-off): NA
(v) Overall cost over the lifetime of the intervention (maximum of 25 years): Moderate

5.27 Water level drawdown (complete and partial)

5.27.1 Description and overview

Water level drawdown is a well-established technique that involves manipulating the water level of a waterbody to expose submerged vegetation to drying (or freezing) conditions, with sufficient desiccation resulting in plant mortality (Figure 19). Minor or rapid water level fluctuations (e.g. up to 1 m) are not likely to control submerged or emergent plants. Since most nuisance plant problems occur in the shallow littoral zone of a waterbody, these areas may be managed by drawdown without having a significant effect on the open water portion of most systems.

The response of submerged vegetation to drawdown can be quite variable and re-growth of problem aquatic plants can be rapid. This method is unlikely to achieve weed eradication unless applied for an extended period of time of months or more and bottom sediments need to be dry to the depth of any plant fragments.

Figure 19 Drawdown in a hydrolake (Photo: R. Wells, NIWA).

5.27.2 Application - *In what situations can the option be applied?*

This method is limited to lakes or ponds that have a dam structure or other mechanism for controlling water level.

The success of a drawdown will depend on the depth, duration, timing and frequency of the water level fluctuation. It also depends on the plant biomass, which can protect plant material in the bottom layers from desiccation. All of these aspects should be considered when devising a program for control. Timing of a drawdown should be planned when the weather is warm and dry and rainfall events are infrequent.

5.27.3 Constraints - *In what situations can the option not be applied?*

This option is not appropriate in constructed wetlands due to possible impacts on beneficial wetland vegetation.

In some systems the base elevation of the outlet or associated subsurface pipe (s) will set the maximum drawdown level for a waterbody (e.g. excavated systems) and may not enable enough water to be removed for drawdown to be effective.

In areas that are managed for ecological values, the incorporation of a 'refuge pool' in deeper water is advocated to support fish and other wildlife while the water level is reduced (Auckland Regional Council 2003). However, this may also provide a refuge for viable weed fragments.

Refilling after drawdown should be avoided at times when cases of **avian botulism** may develop (e.g. water temperatures over 25°C), as the addition of decaying weed material at these times can exacerbate outbreaks.

5.27.4 Requirements - *What other options / practices might be required in conjunction with this option?*

The removal of aquatic plants within this zone by and hand weeding or mechanical means may also benefit this practice. Water level drawdown can provide opportunities for other pond management activities, such as cleaning up of shorelines and for cleaning or repairing structures.

5.27.5 Track record - *Where has this option been successful / unsuccessful?*

Water level drawdown was practiced in NZ hydro-electric lakes for a number of years in the 1980's (Wells et al 2000). While this technique went some way toward controlling nuisance weed growth, the deliberate drawdown for weed control was discontinued primarily on account of the cost (lost hydro-generation potential) and adverse environmental impacts (erosion, slumping). Rapid re-growth of weed often resulted.

While there are many success stories from overseas, information on the use of this method in smaller NZ waterbodies is limited. Water level drawdown to eradicate *egeria* was unsuccessful in a pond beside Lake McLaren, Bay of Plenty (Authors' observations).

5.27.6 Implementation - *Methods employed*

The depth, duration, timing and frequency of the drawdown are critical elements in devising the most beneficial program for control. Success may depend on the slope of a lake or ponds edges and the depth of the littoral zone.

5.27.7 Operation, maintenance, monitoring and reporting

As the response time will not be immediate, the effects of a drawdown will need to be monitored to achieve control.

5.27.8 Practical considerations - *e.g. social issues, access constraints, consent requirements, health and safety concerns*

Response times will vary and not be immediate. During these times areas affected by the drawdown can be aesthetically unpleasing, smell and become a source of dust in windy conditions. If the lake drawdown is carried out during period of drought, water levels may also take a long time to return to acceptable levels.

In some areas the removal of plants along the littoral zone may increase water turbidity due to wind-induced erosion and or re-suspension of sediments. Some systems with complete drawdown have also been noted to experience algal blooms after refilling.

5.27.9 Financial costs

No material costs associated with this option, unless retrofitted drainage is required.

Cost estimate per ha

(i) Start-up / implementation (once-off): Low
(ii) Operation & maintenance (annual): Low
(iii) Monitoring: Low
(iv) Decommissioning, if relevant (once-off): NA
(v) Overall cost over the lifetime of the intervention (maximum of 25 years): Low

5.28 Periodic saline intrusions

5.28.1 Description and overview

Exposure to high salinity conditions may reduce the biomass of, or kill freshwater plants. Plant tolerance to salinity varies according to species-specific mechanisms (e.g. antioxidative enzymes) to protect against oxidative stress under higher salinity (Li et al 2011), and also their history of prior acclimation.

Salinity tolerances for submerged weeds are reported in the literature as between 5 and 8 practical salinity units (PSU) for egeria (Hauenstein and Ramirez 1986), while growth rates of hornwort declined by 78.5% between 0.2 and 3.6 PSU (Li et al 2011) with leaf damage and necrosis reported above 5 PSU (Hinojosa-Garro et al 2008). Another freshwater aquatic weed, *Hydrilla verticillata*, only dominated upper parts of Chesapeake Bay where salinity remained <3 PSU (Shields et al 2012), although it was also reported to tolerate salinity up to 11.6 PSU (Gangstad 1992).

Abrupt salinity incursions are more likely to reduce plant biomass that gradual increases. However, salinity changes may result in proliferation of some salt-tolerant macrophytes, e.g. *Ruppia megacarpa* (Wells et al 2010).

With regard to algal control, artificially flushing lakes or lagoons with seawater has been carried out in many coastal waterbodies to alleviate the symptoms of eutrophication (Suzuki et al 1998; Schallenberg et al 2010). Often these are water bodies that would sometimes open to the sea naturally, although intermittently, and so interventions tend to increase the frequency of openings, rather than exposing an entirely freshwater ecosystem to saltwater. However, the response of these systems to openings is variable, resulting in reduced algal biomass in some, e.g. due to increased flushing of nutrients and phytoplankton, but increased biomass in others, e.g. due to salinity-stratification induced sediment nutrient release (Twomey and Thompson 2001; Schallenberg et al 2010). Moreover, lakes/ponds may become super-saline (through evaporation) if freshwater inflows are not sustained after the lake is opened to the sea.

5.28.2 Application - *In what situations can the option be applied?*

This option may be feasible for those stormwater systems situated close to marine and estuarine areas with the ability to control marine exchanges through outlet gates at higher tidal levels. It may also be possible to pump in seawater to stormwater systems in close proximity to the coast. Judging from tolerance limits in the literature, treatments would need to achieve salinities of approximately 10 PSU (c. 30% seawater) for an extended period of time (weeks to months).

Increased flushing by saline exchanges may also be applicable to lakes/lagoons that are separated from the coast by a barrier that can be breached (e.g. by digging).

5.28.3 Constraints - In *what situations can the option not be applied?*

This option is likely not applicable to situations where opening is likely to cause detrimental effects inside the lake/lagoon or outside in the estuary/coastal area. Maintenance of salinity levels will be difficult in systems receiving large freshwater inflows.

Salinity adjustments are unlikely to effective against emergent weed species. For instance, Alligator weed tolerated a salinity of 10 PSU in flowing water, and is known to survive sea water for 'days' (van Oosterhout 2007).

Increasing salinity may replace one plant problem with another, as, although susceptible freshwater submerged plants may be lost, additional brackish water species may replace them. In Nelson, a stormwater pond managed for model boat enthusiasts experienced on-going issues with the brackish submerged plant, *Ruppia megacarpa* as well as surface algal accumulations of Chlorophyta or filamentous green algae in the genera *Cladophera*, *Ulothrix*, *Ulva* (syn. *Enteromorpha*) and *Spirogyra* (Wells et al 2010).

5.28.4 Requirements - *What other options / practices might be required in conjunction with this option?*

This option would be stand alone, or could be combined with a preceding drawdown to further stress plants.

5.28.5 Track record - *Where has this option been successful / unsuccessful?*

No specific examples of use as an aquatic plant management tool are known. However, the manipulation of salinity to control egeria in the Sacramento-San Joaquin Delta and Upper San Francisco Bay, USA, was not considered feasible (Gartrell 2010).

In NZ both Waituna Lagoon (Southland) and Lake Ellesmere/Te Waihora (Canterbury) are frequently opened artificially to increase lake-sea exchange to reduce inundation of surrounding land and flush water that is high in nutrients, suspended sediment and phytoplankton. However, the efficacy of openings for flushing the lakes differs quite markedly between the two systems, with only a weak response in water quality in Lake Ellesmere and a strong response in Waituna Lagoon, likely driven by differences in morphology and water levels (Schallenberg et al 2010).

5.28.6 Implementation - *Methods employed*

Installation of a gate between the lake/lagoon and the sea may allow control over opening and closing of the lake/lagoon. Digging through the sand/gravel barrier typically allows control over opening only (as closing tends to be a natural process driven by inflows, wind, coastal currents etc.).

Manipulations for plant control would be best carried out in summer, when evaporation may increase salinity levels, and large rainfall events are less likely.

5.28.7 Operation, maintenance, monitoring and reporting

Monitoring of salinity levels (salinity meter) would confirm the treatment time achieved. Water level and water quality monitoring will likely be required to determine when the lake likely needs to be opened. If installed, gates will likely require some maintenance.

5.28.8 Practical considerations *- e.g. social issues, access constraints, consent requirements, health and safety concerns*

Changed salinity will impact on the ecology of the system. Approvals from relevant authorities (local and/or regional) are likely required to artificially open lakes/lagoons to the sea.

5.28.9 Financial costs

Low cost unless pumping of seawater is required.

Cost estimate per ha

(i) Start-up / implementation (once-off): Low
(ii) Operation & maintenance (annual): Low
(iii) Monitoring: Low
(iv) Decommissioning, if relevant (once-off): Low
(v) Overall cost over the lifetime of the intervention (maximum of 25 years): Low

Physical control options for algae

5.29 Substrate capping

5.29.1 Description and overview

Pond and lake sediments accumulate nutrients (particularly phosphorus), which can be, under certain conditions, released into the water column and stimulate algal growth. Anaerobic conditions enhance sediment release of phosphorus, thus sediment lining or capping may be effective at preventing or minimising the release of nutrients in systems where bottom waters become **anoxic**. Sediment lining relies on a passive capping agent, such as sand, gravel or clay, which buries organic matter and sediment and reduces the diffusion of nitrogen and phosphorus into the water column (Klapper 2003; Hickey and Gibbs 2009). Some methods of sediment lining (e.g. with gravel) may also reduce resuspension of sediment and associated nutrients into the water column, by replacing light organic material with mineral materials with higher bulk density.

5.29.2 Application - *In what situations can the option be applied?*

Sediment lining using sand, gravel or clay, requires a thick layer (> 5 cm) of the lining agent, and so is likely applicable only to small systems (due to logistical and financial constraints in large systems).

5.29.3 Constraints - *In what situations can the option not be applied?*

Steep slopes are likely to be difficult to line. Sediment lining using a passive capping agent for the purpose of reducing release of nutrients under anaerobic conditions is unlikely to be effective in systems that do not stratify. Control of external nutrient loading is more appropriate in these cases (Hickey and Gibbs 2009). In shallow systems that are affected by high rates of resuspension capping agents that are made of fine materials are likely to be quickly reworked through the process of sediment focusing (the movement of material from shallower to deeper zones of a lake by water turbulence) or buried into deeper sediment layers (e.g. by bioturbation).

5.29.4 Requirements - *What other options or practices might be required in conjunction with this option?*

Temporary lowering of water levels may assist with installation of sediment lining material.

5.29.5 Track record - *Where has this option been successful or unsuccessful?*

Most examples of sediment capping to control algal growth have used active capping agents (see "Nutrient Inactivation Products" section). However, a well-documented example is the capping of an area of contaminated sediments in Hamilton Harbour, Lake Ontario (Canada), with a layer of c. 35 cm clean medium to coarse sand, which demonstrated a significant reduction in vertical fluxes of soluble reactive phosphorus and other contaminants (Azcue et al 1998).

5.29.6 Implementation - *Methods employed*

Passive capping agents need to be applied in a thick layer (> 5 cm) across the entire area of the pond or lake.

5.29.7 Operation, maintenance, monitoring and reporting

Passive capping agents will become buried by sediment and detritus within a few years of application, and this new sediment layer can release nutrients that can fuel algal growth (Hickey and Gibbs 2009). The rate of burial will be dependent on sediment input, thus new development in the catchment of the stormwater system will likely reduce the efficacy of this option and require new capping material to be applied. For this reason, it is generally accepted that application of capping agents to control internal nutrient loading in lakes should be carried out in conjunction with efforts to reduce external nutrient loading.

5.29.8 Practical considerations - *e.g. social issues, access constraints, consent requirements, health and safety concerns*

Sediment lining with a passive capping agent does not generally require use of potentially toxic products (as with some active capping agents, see "Nutrient Inactivation Products" section). However, passive capping agents will still smother benthic communities and so the ecological effects should likely be assessed on a case-by-case basis.

5.29.9 Financial costs

Costs will be dependent on the size of the area to be treated but lining with sand or gravel is likely to be a relatively low cost option.

(i) Start up: Low
(ii) Operation and maintenance: Low
(iii) Monitoring: NA
(iv) Decommissioning: NA
(v) Overall cost: Low

5.30 Sediment removal

5.30.1 Description and overview

Dredging of bottom sediments can physically remove nutrients from a pond or lake system, but requires knowledge of sediment composition to ensure that sediments are excavated to the correct depth, i.e. partial removal will likely expose sediments with high nutrient content (Hickey and Gibbs 2009). This option can be effective in small systems, but disposal costs can be high, especially if there are high concentrations of chemical contaminants in the sediment (which is likely in stormwater management systems), and contaminants may also be dispersed into lake water when sediments are disturbed during dredging. Furthermore, if external nutrient and sediment loads are not managed concurrently then dredging will likely provide only a short term (c. 2 - 5 years) improvement in water quality (Vanderdoes et al 1992; Faithfull et al 2005; Søndergaard et al 2007). Sediment removal from stormwater management ponds is typically included in best management practice to maintain water storage volume and retention time; it is also likely to be important for controlling algal growth by reducing resuspension of bottom sediments and associated nutrients.

5.30.2 Application - *In what situations can the option be applied?*

In systems where internal nutrient loading from sediments is a major contributor to water column nutrient concentrations, dredging of those sediments will likely be effective at controlling algal growth. Furthermore, dredging may also be effective where reduced depths due to sedimentation are likely to have increased rates of wind-driven resuspension and impact negatively on water quality.

5.30.3 Constraints - *In what situations can the option not be applied?*

In systems where external loading of nutrients and sediment are high, dredging of sediments will likely only be able to provide partial control on algal growth. Consideration should also be given to minimising resuspension of bottom sediments during dredging operations as disturbance of bottom sediments is likely to stimulate algal growth (Figure 20).

Figure 20 Algal bloom in Lake Oranga (University of Waikato) following a dredging operation in 2013.

5.30.4 Requirements - *What other options or practices might be required in conjunction with this option?*

Sediment removal usually requires water level drawdown (partial or complete) to dredge mechanically or with suction dredging.

5.30.5 Track record - *Where has this option been successful or unsuccessful?*

Sediment removal as a lake restoration technique is often undertaken in conjunction with other remediation methods, such as reduction of external loading and biomanipulation. Thus, it is often difficult to determine the impacts/success of each measure individually. However, a study on shallow lakes in the Norfolk Broads (UK) found that dredging could reduce sediment nutrient concentrations; other management methods were also required for controlling algal blooms and restoring macrophytes in the long term (Phillips et al 1999). Sediment removal from shallow lakes on the University of Waikato campus has recently (February 2013) been undertaken in an attempt to remove nutrient-rich sediments, deepen the lakes and reduce the incidence of phytoplankton blooms (Paul and Hamilton 2008); it is still too early to determine the success or otherwise of this restoration effort.

5.30.6 Implementation - *Methods employed*

If the composition of the sediment is unknown then sediment cores will need to be taken from the pond or lake bed to determine the depth of the base substrate (which may or may not be horizontally variable) and the depth of the organic upper layer that would need to be removed (Paul and Hamilton 2008). Excavation could be carried out using earth moving equipment or suction dredging, as appropriate. Consideration needs to be given to dewatering and disposal of dredged material (Figure 21). Detailed descriptions of potential methods and considerations for lake dredging operations are given in Miller (2007).

Figure 21 Dredged spoil pumped from a dredging operation on a lake on the Yangtze River delta (China).

5.30.7 Operation, maintenance, monitoring and reporting

Monitoring of water quality variables, and of sediment depth and composition, both before and after sediment removal, is important to provide quantitative evidence of the efficacy of the option. Regular monitoring is also likely to be required to inform on the need for further sediment removal, particularly if sediment inputs into the pond or lake are high.

5.30.8 Practical considerations - *e.g. social issues, access constraints, consent requirements, health and safety concerns*

As sediment removal is typically carried out as part of routine maintenance of stormwater management systems, the practical considerations of this option should be covered by best management practice documentation. Key issues are likely to include consideration of the disposal of dredged material. This may be particularly important in systems where sediment capping using potentially toxic substances has previously been used, but all spoil should be tested as lake sediments tend to accumulate and enrich contaminants arising from natural and anthropogenic sources.

5.30.9 Financial costs

Costs for sediment removal and disposal will be dependent on a number of factors, including the ease of access for machinery, methodology (e.g. digger vs suction), depth of sediment required to be excavated, and restrictions and requirements for disposal of (potentially contaminated) dredged material. Costs reported in the literature vary widely, e.g. c. $7,000 per ha (Hickey and Gibbs 2009), c. $20, 000 to $50,000 per ha (Miller 2007), c $1,000,000 to $3,000,000 per ha (Healey and Hughes, 2012). Indicative costs for sediment removal for the purpose of de-silting stormwater ponds in the Auckland region (W. Kanz AC pers. comm.) were reported as between $350,000 and $650,000 per ha.

For the overall cost we assume a one-off sediment removal (over the 25-year intervention lifespan considered in this report). If sediment input into the pond is high, for example, due to on-going catchment development, then repeated sediment removal may be required, which will further increase the overall cost.

Cost estimate per ha

(i) Start up: (once-off) Moderate to very high
(ii) Operation and maintenance: NA (unless repeated dredging is required due to high on-going sedimentation rates)
(iii) Monitoring: (not required but recommended to assess effects) Low to Moderate
(iv) Decommissioning: NA
(v) Overall cost: High to very high

5.31 Aeration and artificial destratification

5.31.1 Description and overview

In ponds and lakes with depths greater than a few meters, heating of surface waters during summer induces **stratification**. When waters are stratified, the bottom waters (hypolimnion) can become depleted in oxygen due to microbial activity. The resulting **anoxic** conditions can enhance the release of phosphate and ammonium from bottom sediments. Aeration and artificial destratification are management strategies that aim to regulate nutrient concentrations in the water column by breaking down, or preventing stratification, and oxygenating the entire pond or lake (Hickey and Gibbs 2009). In addition, buoyant **cyanobacteria** tend to be more abundant under stratified conditions and so aeration should create a more turbulent environment less conducive to HAB development. Surface mechanical mixers do not artificially destratify the water column but are designed to prevent buoyant cyanobacteria from accumulating on the surface.

5.31.2 Application - *In what situations can the option be applied?*

This option can be applied in ponds or lakes that stratify and/or those in which algae blooms are typically dominated by buoyant cyanobacteria.

5.31.3 Constraints - *In what situations can the option not be applied?*

In systems that are already well mixed, i.e. shallow water bodies, this option is likely to be unnecessary. Aeration and artificial destratification will only be effective as long as the pond or lake remains mixed, i.e. there will be no long term benefits if the device is removed or switched off. Solar-powered mechanical mixers are unlikely to be applicable in larger systems, which have significant issues with anoxia, as the power required to mix and destratify these systems is likely to be substantial.

5.31.4 Requirements - *What other options or practices might be required in conjunction with this option?*

As artificial aeration does not provide any permanent reduction in internal nutrient loading, it is likely best used in conjunction with management of external nutrient and sediment loading.

5.31.5 Track record - *Where has this option been successful or unsuccessful?*

Mangatangi reservoir, which is 60 m deep, is mixed with one aerator (Hickey and Gibbs 2009). Artificial destratification devices (Figure 22) have recently been installed in Lake Rotoehu, Bay of Plenty (mean depth 8 m), with the aim of mixing the entire water column of the lake and the effects of these devices on water quality are currently being assessed. Artificial mixing in Lake Nieuwe, The Netherlands (mean depth 18 m) has been successful at preventing *Microcystis* (cyanobacteria) blooms (Visser et al 1996). Solar powered circulation has been used for **HAB** control in many lakes in the USA and Canada; case studies of three lakes (mean depth 3 to 7.6 m) indicated that the solar powered circulation was able to strongly suppress **HABs**, although the authors also reported that results were not achieved in other shallower water bodies (Hudnell et al 2010). Solar-powered circulation has also been used in Virginia Lake, Whanganui to suppress cyanobacteria blooms; however, data collected before and after installation is not adequate to quantitatively assess the effect of the device on hydrodynamics or phytoplankton.

Figure 22 Part of an artificial destratifier being installed at Lake Rotoehu, Bay of Plenty. Note screen at bottom intake (left hand side).

5.31.6 Implementation - *Methods employed*

Aerators must be positioned so that the outlets are sufficiently clear of the lake bed to avoid disturbing sediment. Full lake aerators introduce air bubbles along the lake bed, and may create circulation patterns throughout the whole lake. Mixing should be started early enough in spring to prevent the onset of stratification. Mixing can be applied intermittently (if thermal structure is sufficiently well monitored) to reduce operating costs. Generally, an onshore unit (e.g. generator) provides the devices in the lake with power and bubbles.

5.31.7 Operation, maintenance, monitoring and reporting

Aerator or destratifier equipment will require on-going maintenance and a continuous power supply. Monitoring of equipment performance is also likely required to ensure effective operation.

5.31.8 Practical considerations - *e.g. social issues, access constraints, consent requirements, health and safety concerns*

Installation of equipment will likely require vehicle access to the water body. Approval is likely to be required under the Resource Management Act (RMA 1991).

5.31.9 Financial costs

Costs will be site-specific and dependent on size and depth of the lake. However, it is likely to be of moderate (for very small systems) to high cost for most systems, with the majority of the costs involved in the installation (Hickey and Gibbs 2009).

(i) Start up: Moderate-High
(ii) Operation and maintenance: Moderate
(iii) Monitoring: Low
(iv) Decommissioning: (if required) Low-Moderate
(v) Overall cost: Very high

5.32 Ultraviolet light

5.32.1 Description and overview

High levels of ultraviolet (UV) radiation may reduce phytoplankton growth rates (Xenopoulos et al 2002) leading to the suggestion that UV-radiation may be used as an alternative to other treatments to control algae growth in lakes and reservoirs. The advantage of UV-radiation is that no harmful chemicals (such as when treating with algaecides) are added to the water but there are few published studies on the efficacy of this option, particularly in regard to field trials. In laboratory studies, UV-radiation was found to exert a direct control on algae (*Microcystis aeruginosa*) by damaging DNA, and a more indirect control by stimulating production of hydrogen peroxide (H_2O_2) that may also have a toxic effect on the algae (Zamir Bin Alam et al 2001). Other studies indicate that the sensitivity of phytoplankton to UV-radiation may be species-specific, with significantly higher sensitivity of the toxic cyanobacteria, *M. aeruginosa*, compared to the non-toxic green algae, *Chlorella ellipsoidea, C. vulgaris* and *Scenedesmus quadricanda* (Tao et al 2010). Much of the published research into UV-radiation treatment in ponds has focussed on the disinfection of wastewater treatment ponds and effects on bacteria, such as *E. coli* (Craggs et al 2004; USEPA 2006). There are a number of UV-radiation products available for ponds and aquaria in NZ that claim to control algae growth, but there are no published studies on the efficacy of these devices for controlling algae in stormwater or natural ponds or lakes to our knowledge.

5.32.2 Application - *In what situations can the option be applied?*

As there is limited published data on this option it is difficult to determine the likely applications.

5.32.3 Constraints - *In what situations can the option not be applied?*

Unknown.

5.32.4 Requirements - *What other options or practices might be required in conjunction with this option?*

Unknown.

5.32.5 Track record - *Where has this option been successful or unsuccessful?*

We can find no published field studies for this option (for algae control).

5.32.6 Implementation - *Methods employed*

Unknown.

5.32.7 Operation, maintenance, monitoring and reporting

Unknown, but manufacturers and distributors of garden pond UV lights may be able to provide this information.

5.32.8 Practical considerations - *e.g. social issues, access constraints, consent requirements, health and safety concerns*

Unknown

5.32.9 Financial costs

As there is limited information on the applicability of this option for stormwater systems, there is considerable uncertainty around financial costs. However, as UV filter systems for small garden ponds (c. 15,000 to 30,000 litres) can cost c. $2000, the cost per ha for stormwater systems (assuming depth of 1.5 m and therefore volume of 15,000,000 litres) could be c. $200,000. There are also likely to be on-going costs associated with power supply to, and maintenance of, the UV units. Therefore, UV is unlikely to be a low-cost option for algal control in stormwater systems.

(i) Start up: High to Very high
(ii) Operation and maintenance: Moderate
(iii) Monitoring: NA
(iv) Decommissioning: Low
(v) Overall cost: High to Very high

5.33 Ultrasonication

5.33.1 Description and overview

Application of ultrasound as a means to control cyanobacteria has only been considered relatively recently. Ultrasound can disrupt the structure and function of cyanobacterial cells, e.g. by collapsing the gas vacuoles which give cyanobacteria buoyancy, causing them to settle to the lake bottom and reducing access to light. The extent of the damage to the cells is dependent on sound wave frequency, intensity and duration of exposure (reviewed by Rajasekhar et al 2012). Gas vacuoles may be able to regenerate within 20 h after ultrasonication has stopped, although some experiments have suggested that repeated exposure to ultrasound can increase the generation time and thus inhibit cyanobacteria growth more severely (Rajasekhar et al 2012). Long exposure times may result in cell lysis and release of toxins from some cyanobacteria species. Moreover, a concern raised in the recent review by Rajasekhar et al (2012) was that reported studies do not typically focus on the long term effects of sonication on phytoplankton communities, or the wider ecological effects.

5.33.2 Application - *In what situations can the option be applied?*

As ultrasonication damages the gas vacuoles of cyanobacteria this option may be an effective treatment in ponds that have algae blooms composed primarily of those species. Some research has indicated that it may be particularly effective for filamentous cyanobacteria species as the ultrasound may also disrupt the filament structure (Rajasekhar et al 2012).

5.33.3 Constraints - *In what situations can the option not be applied?*

Ultrasonication is less likely to be effective in controlling green algae or diatoms (Ahn et al 2007). Application of ultrasonication in large water bodies will likely require a large number of devices and/or a combination of ultrasounds devices with water pumps to ensure there is adequate exposure to sonication. Application in large systems may therefore be prohibitively expensive.

5.33.4 Requirements - *What other options or practices might be required in conjunction with this option?*

Monitoring of phytoplankton species composition and relative abundance is likely to be necessary to determine the applicability of ultrasonication to a system.

5.33.5 Track record - *Where has this option been successful or unsuccessful?*

Ultrasonication, combined with mixing provided by water pumps, resulted in reduced algal growth, particularly of cyanobacteria, in eutrophic, shallow (2 m depth) ponds in Korea (Ahn et al 2007). Mixed results, ranging from decreases, no change and even increases in algae, have been obtained elsewhere; and tests in UK reservoirs indicated that effects of ultrasonication on algae were inconsistent (Purcell et al 2012).

5.33.6 Implementation - *Methods employed*

This option involves the deployment of ultrasound transducer units at one or more locations within the pond or lake, with the number dependent on the size of the units and the size of the pond. NZ based distributors for ultrasound transducers may be found on the internet.

5.33.7 Operation, maintenance, monitoring and reporting

The ultrasonic transducers require a power supply (mains or solar) and on-going (c. quarterly) maintenance (usually provided by the installer for an annual fee). Monitoring of phytoplankton abundance and composition would be recommended to evaluate the efficacy of the treatment.

5.33.8 Practical considerations - *e.g. social issues, access constraints, consent requirements, health and safety concerns*

Installation would require a secure area to mount the control box and access to a power supply close to the edge of the pond or lake. Also, access to ultrasound transducers in the lake would be required. Approvals from relevant authorities may also be required to install ultrasonic devices (particularly in natural water bodies).

5.33.9 Financial costs

Costs will be dependent on the size of the pond or lake to be treated, but the cost (including installation) of a large unit (which may have a range of up to 500 m) is likely to be c. $8,000-

$10,000. Cost of maintenance and power supply may make this a high cost option in the long term.

(i) Start up: (units and installation) Moderate - High
(ii) Operation and maintenance: Low - Moderate
(iii) Monitoring: Low
(iv) Decommissioning: Low
(v) Overall cost: High

5.34 Wave-attenuation barriers

5.34.1 Description and overview

Wind-driven resuspension of sediments is an important driver of water quality in large, shallow lakes, and may exacerbate the effects of eutrophication. Resuspension of sediments (and organic matter including phytoplankton and particulate nutrients) can result in reduced macrophyte biomass and increased phytoplankton biomass (Hamilton and Mitchell 1996; Ogilvie and Mitchell 1998; Scheffer 2005). Wave-reduction engineering may therefore be effective at reducing sediment resuspension, and improving water quality. Wave-attenuation barriers may be constructed of hard materials, e.g. concrete barrier or soft materials, e.g. nylon curtain (Huang and Liu 2009) (Figure 23). Alternatively, stands of vegetation (e.g. reeds, macrophytes) may reduce wind-driven resuspension (Hamilton and Mitchell 1996; Möller 2006; Falas 2007). (See section 5.7; Macrophyte restoration).

Figure 23 Outer wave-attenuation barrier installed to protect a water treatment plant intake in Lake Taihu, China (Photo: Liancong Luo)

5.34.2 Application - *In what situations can the option be applied?*

This option is applicable in large, shallow, exposed lakes that have high rates of resuspension of bottom sediments.

5.34.3 Constraints - *In what situations can the option not be applied?*

This option is unlikely to be applicable to small ponds or deep lakes, where resuspension of bottom sediments is not an important driver of processes.

5.34.4 Requirements - *What other options or practices might be required in conjunction with this option?*

Installation of barriers may be aided by water level drawdown.

5.34.5 Track record - *Where has this option been successful or unsuccessful?*

There are few published examples for this option. A combination of concrete and nylon barriers were used in Lake Taihu, China, to reduce wind-driven resuspension, and results suggested that the barriers were moderately effective at reducing suspended sediment and nutrient concentrations (Huang and Liu 2009). However, the barriers were not sufficient to reduce phytoplankton biomass, indicating that other measures were also required to restore water quality. See section 5.7; Macrophyte restoration, for details on use of vegetation for reducing wind-driven resuspension.

5.34.6 Implementation - *Methods employed*

Due to the limited information available for this option, and the wide variety of potential barriers (e.g. hard or soft materials, vegetation) it is not possible to give specific details on implementation.

5.34.7 Operation, maintenance, monitoring and reporting

Unknown (see above).

5.34.8 Practical considerations - *e.g. social issues, access constraints, consent requirements, health and safety concerns*

Consideration of the potential aesthetic impact of potentially large structures within the waterbody is advised.

5.34.9 Financial costs

Unknown (see above).

(i) Start up: ?

(ii) Operation and maintenance: ?

(iii) Monitoring: ?

(iv) Decommissioning: ?

(v) Overall cost: ?

5.35 Hydraulic flushing

5.35.1 Description and overview

Flushing a lake with nutrient-poor water reduces the residence time and may increase nutrient export and decrease algal biomass (Klapper 2003). Siphoning may also be used to selectively remove water from a lake; for example, in thermally stratified lakes siphoning of bottom water (known as hypolimnetic withdrawal) may remove nutrient-rich water, reducing nutrients that may become available for algal growth (Hickey and Gibbs 2009). Alternatively, siphoning or skimming of the surface layer, e.g. with a weir, may be used to remove surface algal scums from some systems. This nutrient- or algae-rich water then needs to be discharged downstream, however, which may have negative consequences on receiving environments. Furthermore, an increase in hydraulic flushing will lead to a decrease in water residence time, potentially impacting on the efficacy of the system for stormwater retention.

5.35.2 Application - *In what situations can the option be applied?*

As the main aim of stormwater management systems is typically retention of water to manage flood flows and to improve water quality there are likely to be few situations in which this option can be applied.

5.35.3 Constraints - *In what situations can the option not be applied?*

The option is unlikely to be applicable where the discharge of nutrient- and/or algae- rich water will have negative consequences for downstream systems. Hypolimnetic withdrawal will only be applicable to (deep) lakes that become thermally stratified with high concentrations of nutrients in bottom waters.

5.35.4 Requirements - *What other options or practices might be required in conjunction with this option?*

Unknown.

5.35.5 Track record - *Where has this option been successful or unsuccessful?*

Hypolimnetic withdrawal has been used in a small eutrophic lake (Chain Lake) in British Columbia, Canada to remove phosphorus –rich water and improve water quality (Macdonald et al 2004). Secchi depth was significantly improved in the lake following implementation of hypolimnetic withdrawal, however dissolved oxygen depletion and nutrient enrichment were observed downstream of the withdrawal site. A wall (Figure 24) has been built in Lake Rotoiti to divert nutrient-rich water flowing into the lake through the Ohau channel from Lake Rotorua directly into the Kaituna River (Burns et al 2009). Monitoring subsequent to the wall construction has indicated that the diversion has been successful at diverting the water away from Lake Rotoiti (Hamilton et al 2009).

Figure 24 Ohau diversion wall, Lake Rotoiti. Ohau channel enters on left hand side and is directed towards the Kaituna River outlet towards top of photo (Photo: Andy Bruere).

5.35.6 Implementation - *Methods employed*

This will be highly dependent on the specific method used, e.g. hydraulic flushing with nutrient-poor water requires diversion of a water source to the pond or lake, hypolimnetic withdrawal requires an outflow to be situated below the level of the thermocline (i.e. in the bottom waters).

5.35.7 Operation, maintenance, monitoring and reporting

Dependent on the specific method used.

5.35.8 Practical considerations - *e.g. social issues, access constraints, consent requirements, health and safety concerns*

Discharge of nutrient- and/or algae-rich water to receiving environments may require approval from relevant authorities and/or an assessment of the environmental effects.

5.35.9 Financial costs

The cost will be highly dependent on the specific method used, and this option is typically used in large lakes, rather than ponds so it is difficult to estimate the likely cost where this option to be applicable. However, as most methods will require a highly engineered/constructed solution the cost if likely to be high (Ohau diversion wall was c. $10 million, though this was in a large lake rather than a stormwater management pond).

(i) Start up: Moderate-High
(ii) Operation and maintenance: Moderate
(iii) Monitoring: Low
(iv) Decommissioning: (if required) Moderate-High
(v) Overall cost: High

6.0 Preventative management

6.1 Machinery and materials hygiene

The use of hygiene or quarantine protocols can reduce weed introductions or spread (van Oosterhout 2007). While the development of hygiene protocols is beyond the scope of this report, the main elements of the practise are described here.

Marginal, emergent and submerged aquatic weeds are easily spread by the movement of viable fragments on machinery. Weeds can also be imported to a site with soil, or exported from a contaminated site in spoil. For instance, alligator weed incursions have been associated with soil movements during subdivision developments in Hamilton, NZ. Weed species can also 'hitch-hike' with plantings if these are wild gathered, although the risk with nursery-grown plants should be low.

6.1.1 Application - *In what situations can the option be applied?*

Hygiene protocols should be applied during the construction phase for stormwater systems to prevent problems. For example, care should be taken over the source of substrates to line constructed wetlands. Hygiene would also apply where retrospective earthworks are scheduled, including maintenance de-sedimentation of forebays, wet ponds and wetlands.

Hygiene effort should be guided by the nature and extent of on-site weed issues. For instance, additional efforts should be undertaken where a 'total control' (Table 1) species is present. However, efforts directed at reducing spread of difficult-to-control species, such as alligator weed, would also be cost-effective.

In addition, hygiene should be flagged for the aquatic plant management options that may spread weeds, including Mowing, Mechanical excavation, Suction dredging, Mechanical harvesting or Manual harvesting.

6.1.2 Track record - *Where has this option been successful / unsuccessful?*

There is little information available on the effectiveness of implementing hygiene practices, but numerous examples exist of weed incursions resulting where adequate precautions have not been undertaken (van Oosterhout 2007).

Some information on effectiveness of methods suggests wash-down is a valid method. Visual inspection and hand removal of plant material from the exterior of boats and trailers found that 88% of vegetative macrophyte fragments could be detected, but only 65% of small contaminants such as *Carex* sp. seed (Rothlisberger et al 2010). By contrast, high pressure (1,800 PSI) wash-down increased removal success of small contaminants to 91%, and was much more effective than low pressure washing (4 PSI) (Rothlisberger et al 2010).

6.1.3 Implementation - *Methods employed*

Hygiene protocols for alligator weed infestations in Australia (van Oosterhout 2007) included notifications, operator inductions, signage, restricted access, management of on-site spoil or safe transport and disposal practices, and washdown facilities and procedures for machinery decontamination.

Guidelines for machinery hygiene and clean down procedures (Tyers et al 2004; Vogelzang 2008) included physical methods (e.g. brushing, scraping vacuuming, air compressor, wash-down), chemical methods (e.g. spray-down with detergent), and operational aspects (e.g. designated clean-down areas).

Decontamination of smaller equipment (e.g. rakes, nets) should follow 'Check, clean, dry' principles (MAF Biosecurity 2008) although visual checks will be important for weed seeds.

Protocols developed decontaminating eel fishing nets confirmed a one hour immersion in a 70 g per L solution of NaCl achieved complete kill of six submerged pest plants, but alligator weed and parrots feather survived, probably due to buoyancy and protective waxy cuticle (Matheson et al 2007). Seeds are also likely to be immune to this treatment.

Disposal of contaminated spoil, in the case of low risk weeds, may involve stockpiling and monitoring for weed emergence. For high risk weeds such as alligator weed, deep burial of spoil might be required. Secure disposal sites with burial or silage-type pits were used for alligator weed, with a recommended depth of 3 m, without risk of disturbance (van Oosterhout 2007).

6.2 Nutrient removal via weed harvesting

Theoretically, harvesting of submerged weeds is beneficial due to the removal of nutrients incorporated in plant tissue. However, the benefit to a stormwater system would depend on the amount of weed that can be removed (weed growth rate × biomass × harvesting frequency and efficiency × area of weed bed), compared to the nutrients that are represented in the system (Matheson and Clayton 2002). For instance, nutrients bound in weeds are likely to be a small proportion in systems with high external nutrient loading (Wells and Clayton 2005). A 'nutrient budget' (sources and sinks) for typical stormwater systems in the Auckland region would be required to determine the level of any benefit.

Using values reported in Matheson and Clayton (2002) for the Rotorua lakes, harvesting 1 ha of dense weed bed (41.7 metric tonnes wet weight) could remove 75 kg of nitrogen and 10.4 kg of phosphorus. Harvesting is more likely to remove nutrients derived from sediments as this is the main nutrient source for submerged weeds, with the exception of hornwort (no roots) and free-floating plants, which get most nutrient from the water. Sediment removal is periodically undertaken in stormwater systems. Because submerged plant matter contains lower nutrient concentrations than sediment, harvesting weeds would be much less effective than sediment removal at removing nutrients in stormwater systems.

Harvested submerged plant material is >90% water (Matheson and Clayton 2002). This makes it bulky and expensive to transport. Material could be spread out to dry (or decompose) in the vicinity of stormwater systems as long as there is no opportunity for nutrients to re-enter the aquatic environment. However, the area required for drying material from the Rotorua lakes was estimated at around 10 times the area harvested and there would likely be costs associated with spreading (Matheson and Clayton 2002). The impact of odour from decomposing weed would also need to be taken into account when choosing a drying or stockpiling site. Also, submerged weeds are known to bioaccumulate heavy metals (Xing et al 2013) and this needs to be considered if composting or when disposing of plant material. Potential to use submerged plants for removing heavy metals from eutrophic lakes (**phytoremediation**) is recognised, but factors influencing bioaccumulation are as yet poorly understood (Xing et al 2013).

Where there are doubts about the ensuing benefits, nutrient removal should not be the prime reason for plant harvesting, however removal of submerged biomass after cutting is good practise.

6.3 Avoiding switches to algal dominance

A moderate amount of submerged aquatic vegetation is generally considered beneficial for an aquatic system, as plants may facilitate a number of feedback mechanisms that influence water quality (e.g. wave buffering, sediment settling, bed stabilization, competition with algae). Options that remove substantial amounts of submerged vegetation should also be considered in terms of the risks of a system of a sudden and difficult-to-reverse switch to a turbid, phytoplankton dominated state. Relative to this risk should also be weighed the effect on the system of dense weed beds of invasive pest plants, which can also have significant impacts on water quality and fauna.

Unfortunately, the vulnerability of systems to these switches is poorly understood, but are likely to increase with increasing nutrient status of the system, the proportional influence of the submerged weed target within the system, and environmental conditions (e.g. temperature, meteorological events).

6.4 System design changes

6.4.1 Conversions of wet ponds to wetlands

Reducing the depth of water in wet ponds, or establishing shallow areas for the establishment of emergent plants will decrease available open water habitat. The aim would be the establishment of dense beds of emergent plants that will shade and compete with algae and submerged weeds.

6.4.2 Conversion to rain gardens, dry detention basins, culverts

Conversion to systems that do not support aquatic plants and algae may be one solution to on-going issues with aquatic plants or algae. Rain gardens, rapidly draining, primarily dry detention basins and culverts are unlikely to support aquatic plants or algae. Potential for conversion will be site-specific and larger systems would be less amenable to change.

6.4.3 Surface skimming

Accumulations of small, surface-floating plants (e.g. *Lemna*, *Azolla* species) or buoyant scums of nuisance algae may be skimmed from systems where outlet flows allow their entrainment. For example, a wide, shallow weir or ring drain that draws a large area of surface water could be used in conjunction with temporary water level manipulation to flush surface growths from a system.

6.4.4 Catchment Management

Management practises in the catchment away from the stormwater devices are beyond the scope of this report. However, we note catchment-based practices to reduce sediment and nutrient sources to stormwater are likely to assist in the avoidance of algal and possibly aquatic plant problems. These include regular street sweeping, installation of filters on at-source stormwater intake, avoidance of spills or detergents on street surfaces and increased stormwater infiltration (e.g. porous pavements, grass swales). Many of the algal control options described in this review may only provide short-term solutions to algal bloom problems; concurrent catchment remediation action, targeting the reduction of sediment and nutrient loading, is a critical component of water quality management in aquatic systems.

7.0 Recommendations

Recommendations to improve the utility of this resource are outlined below.

- Hygiene protocols should be developed for machinery, equipment and for materials imported or exported to stormwater system sites to help prevent the spread of biological organisms, including pest plants.
- Consideration should be given to the systematic collation of information for each system that can inform decision-making. For instance, use of this resource requires information on a system (e.g. size, purpose) and the vegetation management problem (e.g. plant type, nature and frequency of issues).
- A number of the control options outlined in this report have not been rigorously tested or proven under the conditions expected in stormwater systems (e.g. ultrasonication, shading by dyes), or they have substantial knowledge gaps (e.g. silver carp, microbial control of algae, mycoherbicide). Future advances in these areas should be noted and addressed in any revision of this report.
- An adaptive management approach to vegetation control activities in stormwater systems, with sufficient monitoring and record keeping by agencies, is likely to assist in addressing some of the uncertainties above. Assessment of the effectiveness of control options would help refine or adapt management practices to suit different stormwater systems.
- Cost estimates in this report are based on generic assumptions and on information from other aquatic environments. Additional information specific to stormwater systems should be sourced where possible to enable refinement of these estimates.
- Development of a costing tool should be explored, to enable more accurate calculation of comparative costs for various options.
- Future development of the decision making framework in this report should consider using electronic media, such as decision tree software or other decision support system options. These tools would aid users in making informed judgements for complex situations by providing a stepwise decision process according to identified criteria and weightings.

8.0 Acknowledgements

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