Media Specification for Stormwater Bioretention Devices

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Media Specification for Stormwater Bioretention Devices

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Executive Summary

Bioretention devices including rain gardens are a Low Impact Design (LID) device primarily used to treat pollutants found in stormwater runoff. The filter media (or fill media) used in bioretention is a critical component to ensure effective bioretention performance. The current design advice in Auckland is a generic "sandy loam, loamy sand, loam, or a loam/sand mix (35-60% sand), with a maximum of 25% clay content" (ARC 2003).

An investigation has been carried out to assess combinations of materials readily available in the Auckland region which create consistent physical and chemical properties to satisfy hydraulic and water quality objectives for stormwater management. The bioretention media should have hydraulic conductivity low enough ensure adequate contact time for pollutant removal mechanisms to be effective, while keeping hydraulic conductivity high enough to minimize (untreated) overflow from water quality volume events. Chemical properties of the media influences the ability to support plant life and influences effluent water quality with potential to balance high hydraulic conductivity/low pollutant-to-media contact time. The investigation process included:

- establishing physical characteristics and performance criteria;
- investigation into available materials;
- particle size distribution (PSD) testing and analysis;
- compaction assessment;
- hydraulic conductivity testing;
- chemical analysis of materials;
- water quality testing.

Multiple types of sand and compost were assessed individually, and then in combinations. The materials and/or product tested during the course of this research were largely selected on the basis of availability. In any section of this report, the information presented is not intended to endorse any particular product or company.

Several international best practice guidelines recommend ranges of aggregate PSD to use as a screening process for achieving desired hydraulic conductivity. The target range of long-term saturated hydraulic conductivity is 12.5 to 150 mm hr^{-1}; however, satisfying this target while considering PSD recommendations and available aggregates proved infeasible. The sand that best satisfied PSD criteria created initial concern that permeability was too high to provide adequate water retention time for broad pollutant removal. Two commercially available bioretention media achieved target hydraulic conductivities with light tamping

compaction. Three fine sands (East Coast Sand [all passing 0.425 mm], Woodhill Black Sand [all passing 0.425 mm], Pumice Sand [all passing 2 mm]), of which the first two do not fit PSD guidelines, showed the greatest potential to satisfy the hydraulic conductivity criteria.

Chemical analyses of the commercial media, composts, and sands were used to assess the potential for pollutant removal and support for plant growth. Phosphorus has likely been added unnecessarily as a fertilizer in both of the commercial bioretention mixes. The addition of compost in 90% sand: 10% compost mixes is the vital component for providing the cation exchange capacity (CEC) required for heavy metal removal, given the near-absence of a clay and silt component (while attempting to adhere to PSD criteria). The importance of sand in these cases is to physically stabilize the system (providing resilience to compaction) and provide an adequate water retention time for the compost components to perform chemical pollutant removal, as well as providing physical filtration to remove contaminants attached to the filtered sediment. However, all media investigated demonstrated potential for phosphorus leaching; all contained at least 10% v/v organic matter as compost.

Based primarily on hydraulic testing results, five media were selected for water quality testing. The selected media were: the two commercial mixes with light tamping compaction, and three 90% sand based mixes (East Coast Sand, Woodhill Black Sand, and Pumice Sand), each blended with 10% bark-based compost and compacted using wetting. Different compaction methods were applied according to media type to approximately provide the hydraulic conductivities identified for assumed adequate media-to-pollutant contact or retention time.

Water quality testing combined simulated water quality storms with concentrated dosing to accelerate media aging in laboratory columns in the absence of plants. Filter media performance was quantified after 0, 5, 10, and 15 years of stormwater loading. Testing focused on dissolved zinc, copper, and phosphorus only as dissolved contaminants are more difficult to remove in many treatment devices, and are also often the bioavailable fraction of the total contaminant (thus driving impacts on aquatic organisms).

Results indicate, for 15 years, the three sand based mixes are capable of removing dissolved copper and dissolved zinc in synthetic stormwater to less than 5 µg/L and 10 µg/L. Mass loads are roughly estimated to be reduced by 60% and 70% respectively. Commercial Mix 2 (CM2) is able to remove dissolved copper and zinc in effluent to 5 µg/L and 13 µg/L, and roughly reduce mass loads by 36% and 46% over 15 years respectively. CM1 displayed initial copper leaching before removing copper, and had inconsistent removal of zinc over 15 years. Over 15 years CM1 was roughly estimated to reduce zinc mass load by 53%, but contribute 15% extra to copper loading.

All five media showed significant phosphorus leaching over the simulated 15 years. CM2 had the highest leaching concentrations (up to 3,200 µg/L), while CM1 had the lowest leaching concentrations (500 µg/L). The three sands were similar in phosphorus leaching levels (approximately 1,200 µg/L). The level of phosphorus leaching should be addressed before media are considered for implementation in phosphorus-sensitive receiving environments.

Two plant species were each planted in two of the sand-based media and one commercial mix for replicated plant growth trials. Biomass accumulation and vigour Carex secta (wet tolerant) and Austrofestuca littoralis (drought tolerant) were measured after 6 months of growth in three bioretention mixes: East Coast Sand + 10% v/v bark-based compost, Pumice Sand + 10% bark-based compost and CM1. Under an as-needed watering regime, plant species grew satisfactorily in all bioretention mixes. Grasses and herbs germinated on all bioretention mixes. The Pumice Sand mix and both CM1 and CM2 stored similar volumes of plant-available water (measured at 10 - 1500 kPa tension). At an installed media depth of 600 mm, approximately 120 - 144 mm of water per bioretention cell unit surface area could be stored by the media tested, whereas at 1000 mm media depth, 200 - 240 mm per unit surface area could be stored.

None of the media mixes tested completely satisfied initial objectives; however, there is evidence to suggest that sand-based, low organic-matter bioretention mixes provide substantial heavy metals' removal capability while enabling plant establishment and growth. Mixes should be tested in the field in combination with organic mulches. Further laboratory work (also followed by field testing), likely including investigation of amendments, would be needed to address phosphorus retention.

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Frequently Used Acronyms and Abbreviations

Descriptor	Abbreviation
Commercially available/proprietary rain garden mix (commercial mix)	CM
Commercially available green-waste based compost	Compost A
Commercially available bark based compost	Compost B
Commercially available peat based compost	Compost C
General (aggregate) all passing X-mm sieve	GAPX
Scoria all passing X-mm sieve	SAPX
Water quality volume	WQV
# years of runoff using a concentrated dose of synthetic stormwater (for water quality testing)	#YCONC
East Coast Sand	ECS
Woodhill Black Sand	WBS
Pumice Sand	PS
90% v/v East Coast Sand + 10% v/v compost B	ECS+
90% v/v Woodhill Black Sand + 10% v/v compost B	WBS+
90% v/v Pumice Sand + 10% v/v compost B	PS+
Proportion of material measured as % of total weight	% w/w
Proportion of material measured as % of total volume	% v/v
Anion exchange capacity	AEC
Cation exchange capacity	CEC
Dissolved organic carbon	DOC

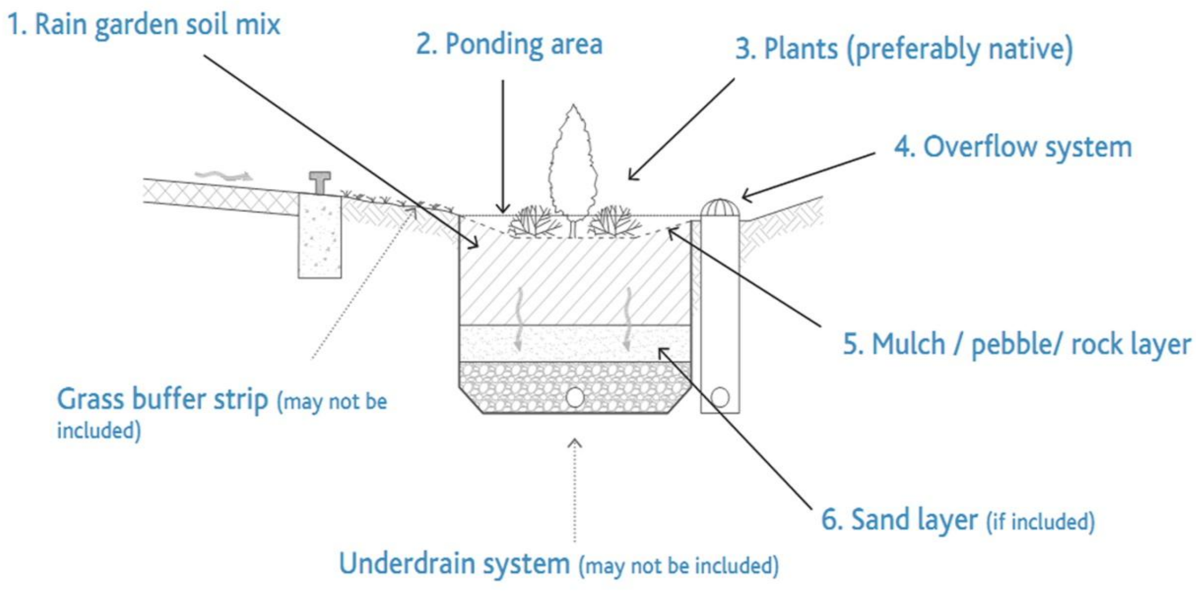
1.0 Introduction

With increasing urban development, the amount of land devoted towards buildings, roads, footpaths, and car parks is growing. These impervious surfaces replace the natural soils which provided substantial stormwater management benefits. The quantity of stormwater runoff from developing areas is increasing, while the quality is decreasing.

The introduction of Low Impact Design (LID) pioneered by Prince George's County, Maryland in the 1990's offered a radical new approach to stormwater management by using on-site, small scale hydrologic controls to mimic the pre-development condition of the site (Prince George's County, 1999). The approach is steadily being adopted worldwide as an effective method of managing stormwater from urban development.

A popular LID device and the subject of this research is the bioretention cell, also known as bioretention, biofilter, bioinfiltration, biocell or rain garden. A bioretention cell is an on-site terrestrial device generally consisting of a drainage layer, transition layer, filter media, mulch layer, and plants (Figure 1).

Figure 1 Bioretention cell components

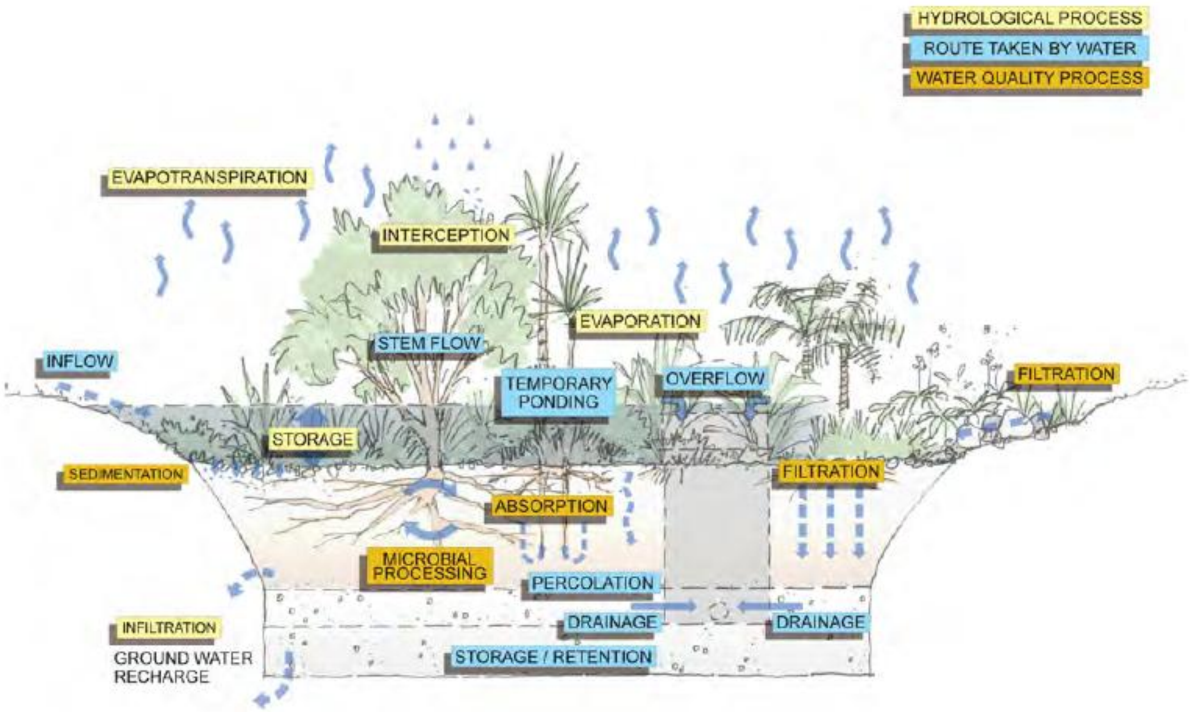


Adapted from Auckland Council Rain Garden Construction Guide (2011)

During and after a rainfall event, runoff flows from a catchment towards the bioretention cell and optimally enters as sheet flow. Ideally, runoff ponds on the surface of the cell and slowly infiltrates into the filter media. An overflow installed above the available ponding surface drains untreated runoff in excess of the ponding depth. Captured runoff is removed by a combination of evapotranspiration, exfiltration to the surrounding soils, and flow through an underdrain to a subsequent system (e.g. reticulated stormwater system, the next operation in a treatment train, or a receiving water).

The bioretention cell uses a combination of filter media, microorganisms and plants to remove pollutants from urban stormwater runoff (Figure 2). Pollutant removal mechanisms include both physical (sedimentation, filtration, adsorption, ion exchange, phytoremediation and volatilization) and biological processes (microbial activity, plant uptake/assimilation, decomposition and thermal attenuation)(Prince George's County, 2007).

Figure 2 Bioretention hydrologic processes and pollutant removal mechanisms



Source: North Shore City Council (NSCC) Bioretention Guidelines (2008a)

While bioretention cells are conventionally designed as a water quality management device, they are also viable as a stormwater flow and volume control device. The field capacity of the filter media stores runoff which is then slowly released through evapotranspiration. Field capacity provides runoff volume reduction regardless of the presence of an underdrain, and is particularly effective for small storm events. Case studies have found bioretention to improve watershed hydrologic characteristics, by reducing peak flows towards predevelopment conditions (Brown & Hunt, 2011a; DeBusk & Wynn, 2011; James & Dymond, 2011).

Other auxiliary benefits of bioretention can include:

- Providing an aesthetically pleasing landscape feature;
- Providing habitat for plants and wildlife;
- Cooling stormwater entering the cell and creating a cooler microclimate surrounding the cell.

1.1 Bioretention filter media specifications

Most of a bioretention cell's potential benefits rely on characteristics of the filter media. The filter media should:

1. Allow adequate infiltration and permeability;
2. Have the necessary chemical properties to facilitate pollutant removal;
3. Allow adequate contact time with the stormwater for pollutant removal to take place;
4. Provide adequate nutrients, aeration, moisture storage, and physical support for plants, allowing plant root extension;
5. Remain stable over a relatively long term without shrinking, compacting or structurally collapsing.

1.2 Design and construction issues

Poor construction and design errors may limit the effectiveness of bioretention in both hydrology and water quality aspects. A major reason for failure of bioretention cells stem from improper design or poor construction techniques (Warynski & Hunt, 2011). Hydraulic failure of bioretention media can be due to:

- Incorrect media specification, where the media has incorrect physical/chemical properties for removing targeted pollutants.
- Incorrect media specification, where the media may have high clay content or extremely fine particles, and vulnerability to compaction which cause inadequate drainage and over-extended ponding.
- Incorrect compaction, often resulting from poor compaction specifications or using media vulnerable to compaction. The media is either under-compacted and too loose resulting in low contact time, or over-compacted and too dense resulting in inadequate drainage and over-extended ponding.

- Clogging, where excessive sediment loads restrict the pores of the media, hindering infiltration and causing inadequate drainage and over-extended ponding. Clogging most commonly occurs at the surface as crusting, capping, or sealing. Sediment from unstable catchments, catchments with active construction, or fine particles within the filter media contribute to clogging.

Jackson (1990) sums up the importance of technical specification during construction: "Clear specifications and a consistent understanding of the intent of specifications by all parties leads to a project of higher quality. The specifications also protect the right of the contractor to choose methods of construction and provide assurance that the owner will receive the required performance of the finished product". Furthermore, "The prompt clarification of ambiguities and the fair resolution of discrepancies concerning technical specifications leads to projects with fewer disputes, resulting in a reduced need for arbitration and litigation and thus lower overhead costs for both owners and contractors".

Design and construction flaws can produce underperforming bioretention cells. Brown & Hunt (2011b) in North Carolina encountered volumetrically undersized bioretention cells, resulting in overflows becoming a frequent occurrence and significantly reduced performance. After enlarging the bioretention cells to the correct size, an additional 25% of annual runoff was treated, resulting in improved water quality treatment, reduction of peak outflow, and reduced duration of high outflow. As an on-going interest in inaccurately constructed bioretention cells in North Carolina, Warynski & Hunt (2011) performed an inspection of 20 bioretention cells throughout the state. They found 82% of bioretention cell filter media having incorrect particle size distributions (PSD), and 44% of bioretention cells having incorrect permeabilities. Furthermore, 50% of bioretention cells were undersized. These findings clearly show a failure somewhere between the design and construction stages of bioretention cells, which in effect lead to poor bioretention performance and money wasted.

Careful installation of the filter media may be required to ensure the proper design mechanisms for bioretention are achieved. However, constructed bioretention cells often deviate from design manual and guideline recommendations (Stander et al. 2010). Reasons for out-of-protocol construction include (Stander et al. 2010): logistical problems in locating specific filter media, high cost of filter media, variations among design manuals and guidelines, and attempts to modify bioretention design for site specific conditions. With construction problems for new bioretention designs appearing to be highly prevalent, Stander et al. (2010) recommends "bench-scale" testing in order to evaluate the performance of new bioretention media designs before they are built on a full scale.

However, bench scale testing may still be susceptible to construction problems when implemented on a full scale.

1.3 The role of plants

Bioretention needs vigorous plant cover to maximize performance as plants provide, or contribute to stormwater management functions including:

- Sedimentation and erosion control. Dense foliage physically protects the substrate surface from erosion and slows stormwater velocity; both help minimise surface re-suspension of deposited sediment. Foliage also impedes movement of floating materials (litter and some organic mulches) into overflows.
- Microbial processes. Plants provide organic substrates on which many microbial processes are based, particularly in the rhizosphere (around roots) and decomposing leaf litter.
- Nitrogen and phosphorus removal. Plants extract these macro-nutrients when actively growing; decomposing leaves and roots gradually release these but at a rate that can be re-used by the plants (rather than leached).
- Metal removal. Soluble metals are taken up by plants during active growth periods and incorporated into leaves and roots. High biomass is usually associated with greater metal removal.
- Stormwater volume attenuation. Evapotranspiration creates air-filled pore volume within the media to store stormwater, therefore contributing to the volume that can be treated before overflow occurs.

Plants play an important role in maintaining adequate surface infiltration and permeability of bioretention media. Infiltration is maintained by shoot and root growth & active invertebrate activity (incorporating dead plant material); and the potential for compaction is reduced by foliage that provided both physical surface protection, and a visual barrier to entry by people and vehicles. A network of roots helps maintain resilience to compaction from vibration, and decomposing roots refresh macropores.

Plants also provide the bulk of the amenity values provided by bioretention. There is potential conflict between the amenity and pollutant removal functions of bioretention. Rapid, lush plant growth is generally seen as desirable (high amenity) and is achieved in the landscaping industry by providing abundant levels of the macro-nutrients nitrogen and phosphorus, both of which are also potential surface water pollutants that bioretention can be designed to attenuate. High fertility is achieved in the landscaping industry by using organic-enriched substrates (i.e., by adding composts to soil), inorganic fertilisers and/or organic mulches. Fertilisers may be needed in standard landscaping because they receive little surface runoff and irrigation uses roof-runoff or potable supply so is low in nitrogen and phosphorus. In contrast, bioretention is designed to receive water from a catchment 20 to 50 times larger than the bioretention (2 to 5% of catchment), and this stormwater generally includes sediment and nutrients washed from urban surfaces. Bioretention is therefore considered to be self-fertilising, so not requiring either high organic matter levels or supplemental fertilisation once plants are established.

1.4 Research objectives and scope

This research is to support Auckland Council's Stormwater Technical Services objectives of providing technical guidance for design of stormwater management devices based on international best practice adapted for local conditions.

Bioretention filter media needs to balance five major design criteria:

- Have a high enough hydraulic conductivity to allow for surface infiltration of stormwater meeting a specified time of water drawdown from a maximum ponding depth to prevent extended water ponding (i.e. minimum hydraulic conductivity).
- Have a low enough hydraulic conductivity to allow for stormwater to be retained in the media for a sufficient contact time to allow for pollutant removal mechanisms to operate (i.e. maximum hydraulic conductivity).
- Have chemical composition to remove pollutants.
- Provide plants with required nutrients and water to allow for sustained long term growth.
- Be structurally stable and maintain even flow through media (avoiding preferential flow).

The aim of the first stage of research was to assess how candidate bioretention filter materials react to different mixing and compaction treatments and how their performance compares to criteria established from a literature review. Important assessment criteria includes material that:

- Meets the design criteria.
- Is readily available from New Zealand sources (preferably in the Auckland region).
- Can be consistently supplied with adequate quality control.

The aim of the second stage was to carry forward the best candidate media from the first stage and conduct water quality tests. Water quality tests aim to provide estimation into the pollutant removal ability of the media, as well as developing an understanding of how the pollutants are being removed. Several of the mixes tested for water quality performance were also subjected to plant growth trials in a shaded glasshouse.

The scope of the investigation was limited to field visits to materials' suppliers and laboratory testing. The scope did not include field verification of laboratory findings, or recommendations for appropriate plants. The second stage of research included laboratory water quality testing with synthetic stormwater and plant trials.

2.0 Local and international design guidelines

Bioretention design guidance from around the world was reviewed to establish desirable physical characteristics and related hydraulic properties of the fill media.

The current design guideline manual for stormwater treatment devices in Auckland is known as Auckland Regional Council's Technical Publication 10 (ARC TP10), published in 2003 (Auckland Regional Council, 2003). TP10 chapter 7 "Filtration design, construction and maintenance" has a section devoted to bioretention (called rain gardens). The chapter discusses the design approach in terms of sizing, general composition of filter media, plant material, construction advice, and maintenance required.

In New Zealand, as well as Auckland Regional Council's TP10, the former North Shore City Council published the Stormwater Management Practice Notes on bioretention (North Shore City, 2008b) and former Waitakere City Council published Stormwater Solutions for Residential Sites including a section on bioretention (Waitakere City Council, 2004). Hamilton also has booklet information on bioretention (Hamilton City Council, 2006). On an international scale, guidelines, specifications or standards tend to be published by individual cities and/or states. Some examples include Prince George's County in Maryland (Prince George's County, 2007), Seattle and Puget Sound in Washington State (Puget Sound Partnership, 2009; Seattle Public Utilities, 2008), North Carolina (Hunt and Lord 2006), California (CASQA, 2003) and Melbourne (FAWB, 2009a).

2.1 Bioretention filter media composition

The composition of the filter media is fundamental to the success of the bioretention cell. A summary of recommended bioretention filter media composition is presented in Table 1.

Table 1 Recommended bioretention filter media mixes from worldwide sources[1]

Guideline	Aggregate	Organic	Note
Auckland Regional Council (2003), Waitakere City Council (2004)	Sandy loam, loamy sand, loam, loam/sand mix (35 - 60% v/v sand)	Not specified	Clay content < 25% v/v
Prince George's County, Maryland (2007)	50 - 60% v/v sand	20 - 30% v/v well aged leaf compost, 20 - 30% v/v topsoil2	Clay content < 5% v/v
The SUDS manual (Woods-Ballard et al. 2007)	35 - 60% v/v sand, 30 - 50% v/v silt	0 - 4% v/v organic matter	10 - 25% v/v clay content
Facility for Advanced Water Biofiltration (FAWB, 2009a)	Washed, well graded sand with specified PSD band	3% w/w organic material	Clay content < 3% w/w, top 100 mm to be ameliorated with organic matter and fertilizer
Seattle Public Utilities (2008)	60 - 65% v/v mineral aggregate, PSD limit ("clean sand" with 2 - 5% passing #200 sieve), U3 $\geq$ 4	35 - 40% v/v fine compost which has > 40% w/w organic matter content	
Puget Sound Partnership (2009)		40% v/v compost, or 8 - 10% w/w organic matter	
North Carolina Cooperative Extension Service (Hunt & Lord 2006)	85 - 88% v/v washed medium sand4	3 - 5% v/v organic matter	8 - 12% v/v silt and clay

Guideline	Aggregate	Organic	Note
City of Austin (2011)	70 - 80% v/v concrete sand5	20 - 30% v/v screened bulk topsoil2	70 - 90% sand content, 3 - 10% clay content, silt and clay content < 27% w/w. Warning not to use sandy loam (“red death”).6

1. % v/v is percent by volume; % w/w is percent by weight (mass).
2. “Topsoil” is a non-technical term for the upper or outmost layer of soil, however there is no technical standard for topsoil.
3. U, Coefficient of Uniformity = D_{60}/D_{10}, where D_{60} is particle diameter at 60% passing and D_{10} is particle diameter at 10% passing.
4. A specific definition for “medium sand” was not identified. ASTM D2487-10 classifies coarse-grained sands as those with ≥ 50% retained on the (USA) No. 200 sieve (75 ∝m) and ≥ 50% of coarse fraction passing the No. 4 sieve (4.76 mm). Clean sands contain < 5% fines. Fine-grained soils are silts and clays whereby > 50% passes the No. 200 sieve.
5. Concrete sand is described by ASTMD2487-10 as coarse sand that is retained by a (USA) No. 10 sieve (2.00 mm)
6. “Red death” is commercially available fill material in Austin marketed as sandy loam.

Traditionally, filter media has been a homogeneous mix of sand, topsoil and organic matter. Prince George’s County, Maryland initially selected three textural soil classifications (Figure 3) based on the minimum United States Department of Agriculture (USDA) natural soil infiltration practices in Maryland: loamy sand, sandy loam, and loam (Prince George’s County, 1999). After experiencing construction failures, a revised mix of 50 to 60% concrete sand, 20% to 30% leaf-based compost and 20 to 30% topsoil by volume was adopted [this is also the most recent Maryland mix found in the Bioretention Manual (Prince George’s County, 2007)]. “Topsoil” is a nontechnical specification, which may contain anything from silty loam to sandy clay loam (Figure 3). Loamy sand can contain up to 20% clay. Clar et al. (2007) considers the design of a revised mix as a knee-jerk reaction, and blames the early failures on lack of onsite soil testing and the loam textural classification, which may contain up to 33% of infiltration-hindering clay[1]. Many other guidelines (such as CASQA and ARC [2003]) tend to be based on the old 1999 Prince George’s County, Maryland’s recommendation of using a textural soil classification. On the other hand, the city of Austin’s 2011 guideline specifically carries a warning not to use commercially available fill material

[1] Clay is a generic term that includes various mineralogies; not all clays hinder infiltration. However, the statement in the text is stated in a manner to reflect the original reference.

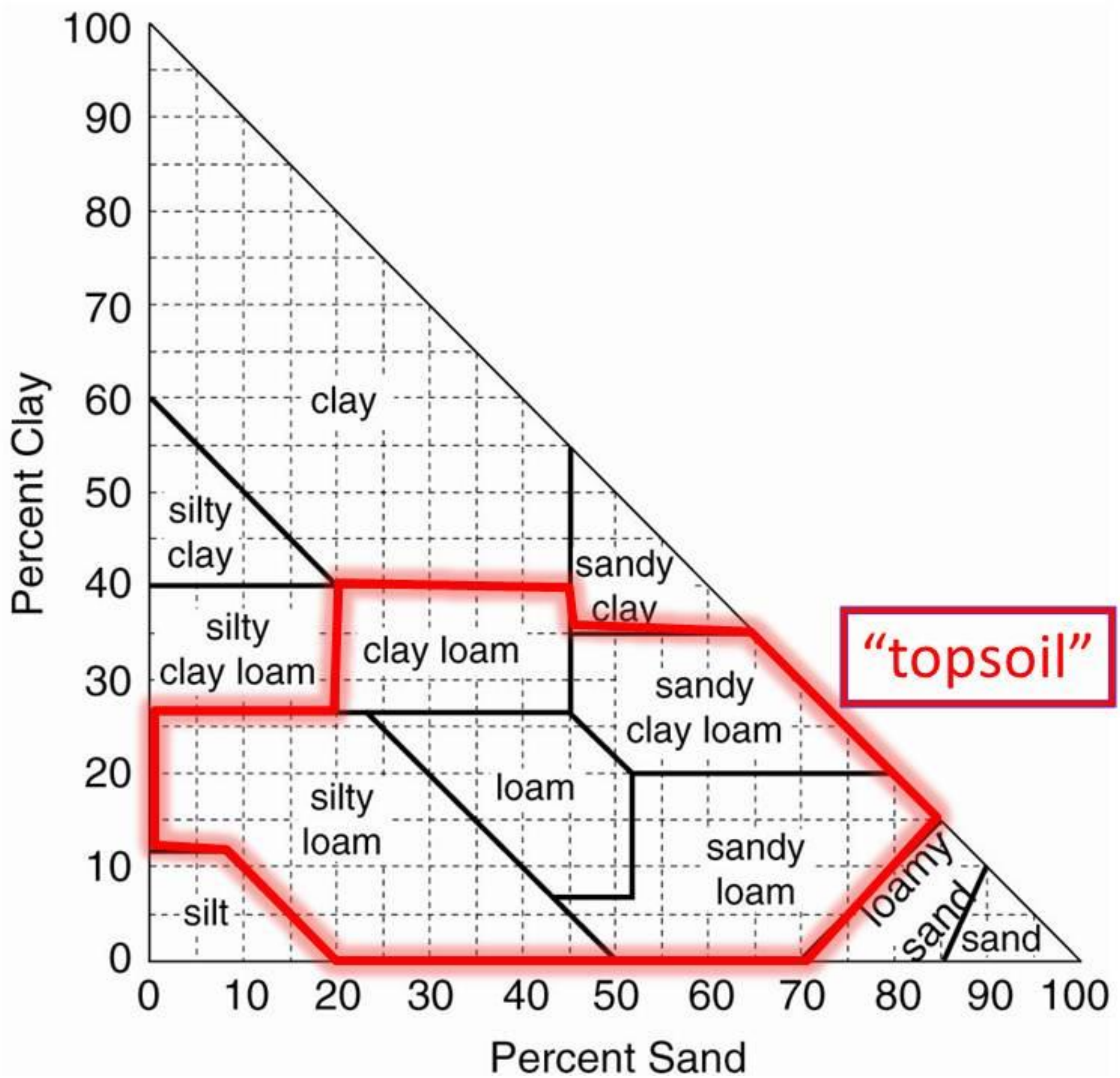
marketed as “sandy loam”, which is deemed to be infertile and have poor drainage characteristics (and is also known colloquially as “red death”).

Rather than relying on textural classification, the Facility for Advanced Water Biofiltration (FAWB) from Monash University in Melbourne uses a particle size distribution (PSD) guideline to help achieve the required hydraulic conductivity. Seattle Public Utilities (SPU) and Puget Sound Partnership in Washington State also classify appropriate filter media using PSDs.

All of the guidelines reviewed limit the amount of clay in the filter media. Clay can contribute to low conductivity or clogging, which can lead to overextended ponding times or premature overflows, both of which count as failure of the bioretention cell. However, saturated hydraulic conductivity of clays are dependent on the mineralogy and compaction state which vary throughout different locations, so careful consideration of maximum clay contents should be made for different localities. The minimum percentage of a specific clay product (used in American baseball fields, and thus has tight quality control) is specified in North Carolina (Hunt and Lord 2006). Some clays greatly enhance pollutant removal capacity of substrates, particularly those high in aluminium or iron oxides. Clay can also boost water retention, which contributes to plant-available water and runoff volume attenuation.

Clay mineralogies in Auckland vary enormously. Iron- and aluminium-rich clays on younger basalt volcanoes contain allophane, gibbsite, goethite and haematite, are permeable, friable and bright red or brown. They contrast with low-permeability clays dominated by layer-silicate clays (kaolinite, halloysite and gibbsite) found on sedimentary rocks and have dispersible horizons (layers) that are susceptible to compaction damage when wet (Malloy 2008). The former are classified as Oxidic and Granular Soils in the NZ Soils Classification, the latter generally Ultic and Gley Soils (Malloy 2008). Granular soils from the Pukekohe area, used to create cricket wickets, were investigated for use in bioretention, however, testing in the late 2000s showed very high soluble (Olsen) phosphate concentrations were present (Landcare Research unpublished data), indicating that they would not be suitable for use as a stormwater treatment media (as phosphorus can be a pollutant of concern in runoff). Other than the Pukekohe area, Granular Soils are found on pockets of hydro-thermally altered sandstone within the Waitemata Formation in Auckland - the pink-red colouration created by haematite clays (Ross 2007). The Auckland region also contains silt-textured allophanic soils, developed from remnant pockets of air-fall tephra (volcanic ash) and older tephra reworked by water or wind-sorting (Ross 2007) which are also high in aluminium (allophane, also imogolite or ferrihydrite) and also bind phosphorus (increase Anion Exchange Capacity). A commercially-available, highly consistent resource has not been identified.

Figure 3 USDA textural classification of natural soils.* All types within the red outline are considered "topsoil".



*Adapted from USDA (www.soils.usda.gov, accessed April 2012).

The organic proportion is highly variable across the guidelines, from as low as 3% up to 40% v/v. On-going research by Clark & Pitt (2011) reveal both cation exchange capacity (CEC) and a high organic matter content to be important when specifying media to remove a wide range of metallic pollutants. Furthermore, removal of the dissolved fraction of metals requires the media to be slightly acid to slightly alkaline due to increased solubility of metals at moderately acidic pHs (< 5.5) and plant phosphate deficiency likely in moderately alkaline pHs. However, the removal of phosphorus is at an optimum when the media has a higher pH and lower organic matter content. The materials' specification (Table 1) from North Carolina significantly restricts organic content, but provides for water and nutrient retention through a very tight specification for a particular type of clay (Hunt and Lord 2006).

Seattle Public Utilities (SPU) stands out amongst the guidelines summarized in Table 1 for a relatively larger fraction of organic matter addition. SPU's mix relies on the compost component to reduce the surface infiltration rate to an acceptable level (25 - 305 mm hr^{-1}), for pollutant removal, and to promote robust plant growth (in part through water storage capacity). The local industry relies on a tight compost specification with compliant suppliers. The actual organic content of the composts meeting their spec is usually 5 - 8% w/w (Colwell, personal communication, 2012). A particle size distribution for the compost is also specified (see Section 2.5).

The predicament of trying to balance these characteristics points to a need for greater understanding of the interaction between water and soil chemistry in order to design bioretention cells for effective pollutant treatment.

2.2 Media depth

Table 2 summarizes recommended media depths from various guidelines and literature in chronological order. Media depth recommendations have been decreasing from over 1,000 mm in 1999 to around 500 mm in the late 2000s, as research has found most pollutants are removed in the top 20% (i.e. 200 mm) of the filter media (see section 1.6.1). Media depths are generally in multiples of 'feet', here converted to mm.

Less media volume (from shallower media depth) results in less organic matter overall and therefore less chance of nutrient leaching. Shallower media depth also reduces the material and building cost of bioretention cells and allows for construction in areas with shallow water tables. However, shallow rooting volume can also restrict plant selection to shorter vegetation as trees are deeper rooting than herbaceous groundcovers.

Table 2 Recommended media depths (in chronological order by source).

Source	Media Depth
USEPA Storm Water Technology Fact Sheet Bioretention (USEPA, 1999)	1,219 mm
Prince George's Country, Maryland (1999)	610 - 1,219 mm
USEPA Stormwater Best Management Practice Design Guide Volume 2, Vegetative Biofilters (USEPA 2004)	610 - 1,219 mm
North Carolina Cooperative Extension Service (Hunt and Lord 2006)	610 - 1,219 mm
Prince George's Country, Maryland (2007)	762 - 1,219 mm
The SUDS manual (Woods-Ballard et al., 2007)	1000 mm (min)
Washington State University (Hinman 2007)	305 - 610 mm
FAWB (2009b)	300 - 800 mm
Washington State University (Hinman 2009)	457 mm (min)
Auckland Council Rain Garden Construction Guide (2011)	700 mm (min)
City of Austin (2011)	457 mm

2.3 Ponding depth

Table 3 summarizes recommended ponding depths from various sources in chronological order. Allowable maximum ponding depth seems to be increasing. Increasing ponding depth effectively allows for a reduction in bioretention cell footprint whilst maintaining control over the same volume of stormwater.

Ponding depth should be linked to an appropriate hydraulic conductivity, otherwise extended water ponding time can occur, impacting plant health and species selection. Ideally, plants should have some foliage above the maximum ponding depth to lower the potential for sediment impact on leaves. Ponding duration over 3 - 4.5 days may allow breeding of mosquitoes and other disease carriers, creating public health risk (Center for Disease Control, undated; Roy-Poirier et al., 2010; Virginia Department of Health, undated), or at least the potential for poor public acceptance. Conversely, inadequate ponding depth (or hydraulic conductivity) precludes runoff capture and treatment with a larger proportion of runoff bypassing the system.

It is assumed that the ponding depths in Table 3 have been identified primarily as they relate to water quality treatment, overall "health" of the bioretention system, and public safety. Hunt et al. (2012) suggest that temporary ponding greater than 300 mm may be beneficial for peak flow control, while the water quality volume should still be restricted to a 300 mm maximum.

Table 3 Recommended ponding depths (in chronological order by source)

Source	Ponding Depth
USEPA Storm Water Technology Fact Sheet Bioretention (USEPA, 1999)	152 mm (max)
Prince George's County, Maryland (1999)	152 mm (max)
Auckland Regional Council TP10 (2003)	220 mm (max)
USEPA Stormwater Best Management Practice Design Guide Volume 2, Vegetative Biofilters (USEPA, 2004)	152 - 305 mm
University of Wisconsin-Madison (2006)	457 mm (max)
Prince George's County, Maryland (2007)	152 - 305 mm
The SUDS manual (Woods-Ballard et al. 2007)	150 mm (max)
Washington State University (2007)	152 - 305 mm
FAWB (2009b)	100 - 300 mm
North Carolina (Brown & Hunt, 2011b)	300 mm (max)

2.4 Hydraulic performance

The difference between hydraulic conductivity and surface infiltration is often confused due to their similar units of distance over a time period (usually mm hr^{-1}). Fredlund & Rahardjo (1993) differentiate the two terms by designating hydraulic conductivity as "the ease of which water can move through pore spaces", and infiltration as "the process by which water enters the soil". Horton (1933) described infiltration capacity as "the maximum rate water can enter the soil at any particular point under a given set of conditions". Factors affecting hydraulic conductivity of a given soil include PSD, porosity, grain angularity, degree of compaction, and presence of clay particles (Bell, 1998).

Table 4 shows recommended saturated hydraulic conductivity of bioretention filter media from various international guidelines. The minimum hydraulic conductivity out of all the guidelines is 12.5 mm hr^{-1}, which is equivalent to 24 hour drawdown from 300 mm ponding depth. Most of the rates in Table 5 are presented as minimums to prevent extended ponding water, and hence the minimum hydraulic conductivities should not be regarded as a target.

FAWB (2009a) recognised potential issues at several ranges of hydraulic conductivities. Plant survival and pollutant removal potential would be at risk in the 600 to 800 mm hr^{-1} range for plants in Melbourne (very high summer evapotranspiration rates). Plants could fail to establish in the 350 to 600 mm hr^{-1} range. This can be ameliorated by supplemental irrigation or timing planting (to periods without drought stress, i.e autumn or winter). The large amount of required filter area to treat stormwater volumes would be an issue at hydraulic conductivities below 100 mm hr^{-1}. While keeping in mind most hydraulic conductivities in Table 4 are minimums, all publications except the FAWB are in the range for which the FAWB considers the large filter area required to be an issue. Quantifying actual hydraulic conductivity for available or typical media may improve device sizing procedures. Where hydraulic conductivity is greater than minimum requirements, the calculated footprint for a bioretention cell for a given drainage area would reduce.

While hydraulic conductivity initially declines as the filter media is compacted (see section 5.5), FAWB (2009a) and Barret et al. (2011) found it often recovers back to the design value over time as increased plant root growth counters the effects of compaction and clogging.

Table 4 Recommended hydraulic conductivity of bioretention filter media

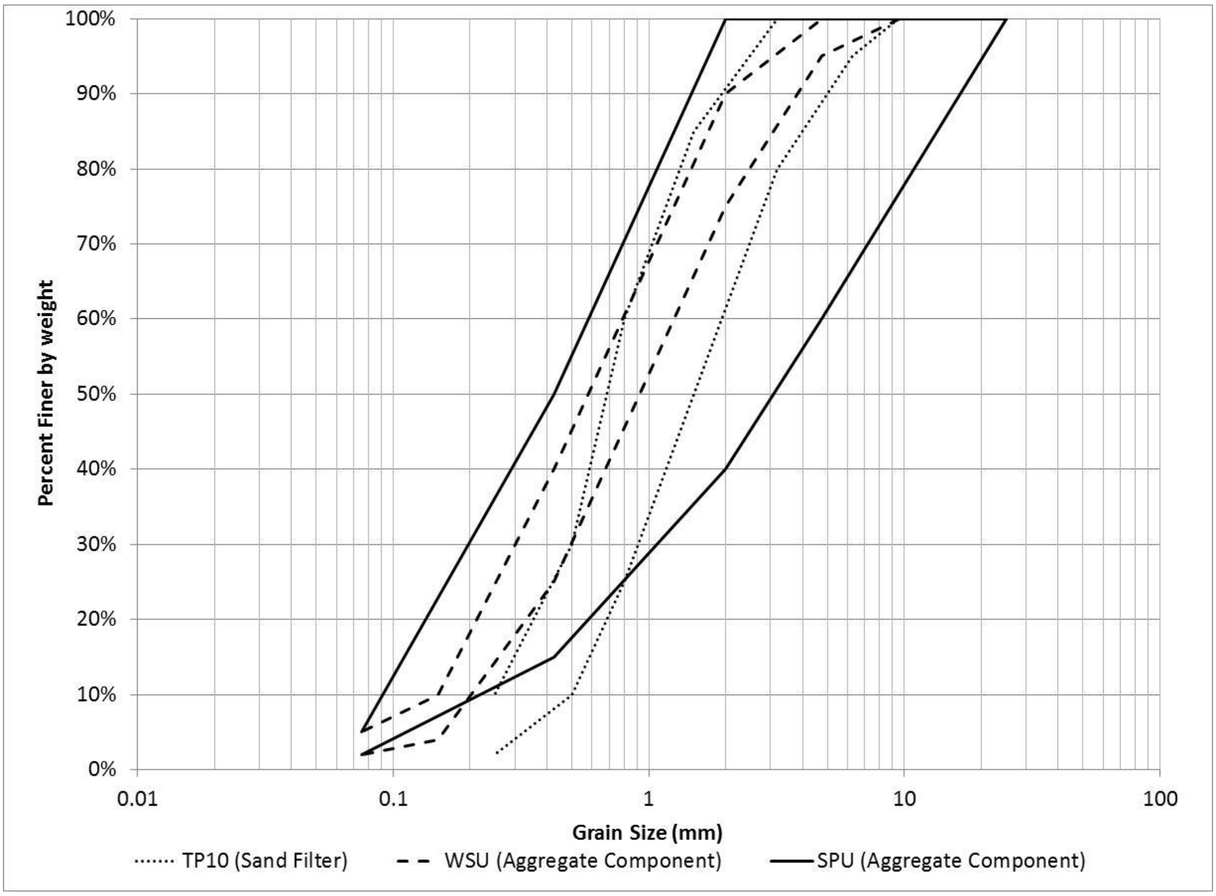
Publication	Hydraulic Conductivity
Auckland Council Rain Garden Construction Guide (2011)	12.5 mm hr^{-1} (min)
California Bioretention TC-32 (CASQA, 2003)	12.5 mm hr^{-1} (min)
City of Austin (2011)	50.8 mm hr^{-1} (min)
USEPA (2004)	12.7 mm hr^{-1} (min)
FAWB (2009b)	100 - 300 mm hr^{-1} (temperate climates) 100 - 500 mm hr^{-1} (tropical climates)
Prince George's County, Maryland (2007)	12.7 mm hr^{-1} (min)
The SUDS manual (Woods-Ballard et al. 2007)	12.6 mm hr^{-1}
North Carolina Cooperative Extension Service (Hunt and Lord 2006)	25.4 mm hr^{-1} (for nitrogen removal) 50.8 mm hr^{-1} (for phosphorus, metal and other pollutant removal)
Puget Sound Partnership (2009) Seattle Public Utilities (2011)	25.4 - 305 mm hr^{-1}

2.5 Particle Size Distribution

Particle size distribution (PSD) is used as a gauge of a potential filter media's hydraulic performance of a filter media in several international guidelines. Maximum and minimum PSD guidelines are provided by Seattle Public Utilities (SPU) and Washington State University (WSU). A "banded" PSD guideline from the Melbourne's Facility for Advanced Water Biofiltration (FAWB 2009 a, b) was also identified. The maximum and minimum PSD guideline for sand filters from Auckland Regional Council's TP10 (ARC 2003) is the most relevant local guidance. These guidelines are presented in graphical form (Figures 4-6).

Particle Size Distribution (PSD) may be a useful gauge of the potential hydraulic performance of a filter media, but it should not be used to replace hydraulic conductivity testing. As well as meeting gradation limits, media should be well-graded over the entire range to avoid structural collapse due to particle migration (FAWB, 2009a).

Figure 4 Aggregate PSD limits for assessing candidate bioretention filter media*



*TP10 = Technical Publication 10 (ARC 2003) WSU = Washington State University (Hinman 2009)
SPU = Seattle Public Utilities (2008)

Figure 5 Aggregate PSD bands for candidate bioretention filter media according to FAWB (2009 a, b)

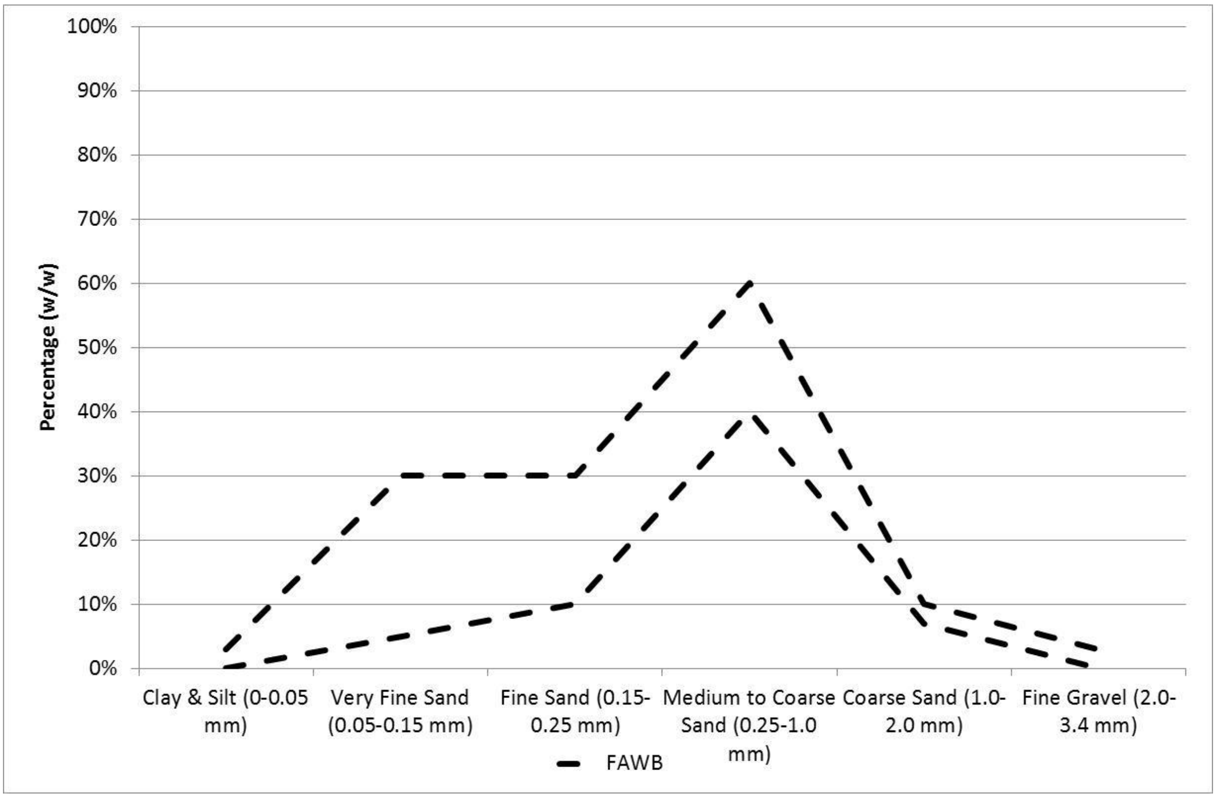
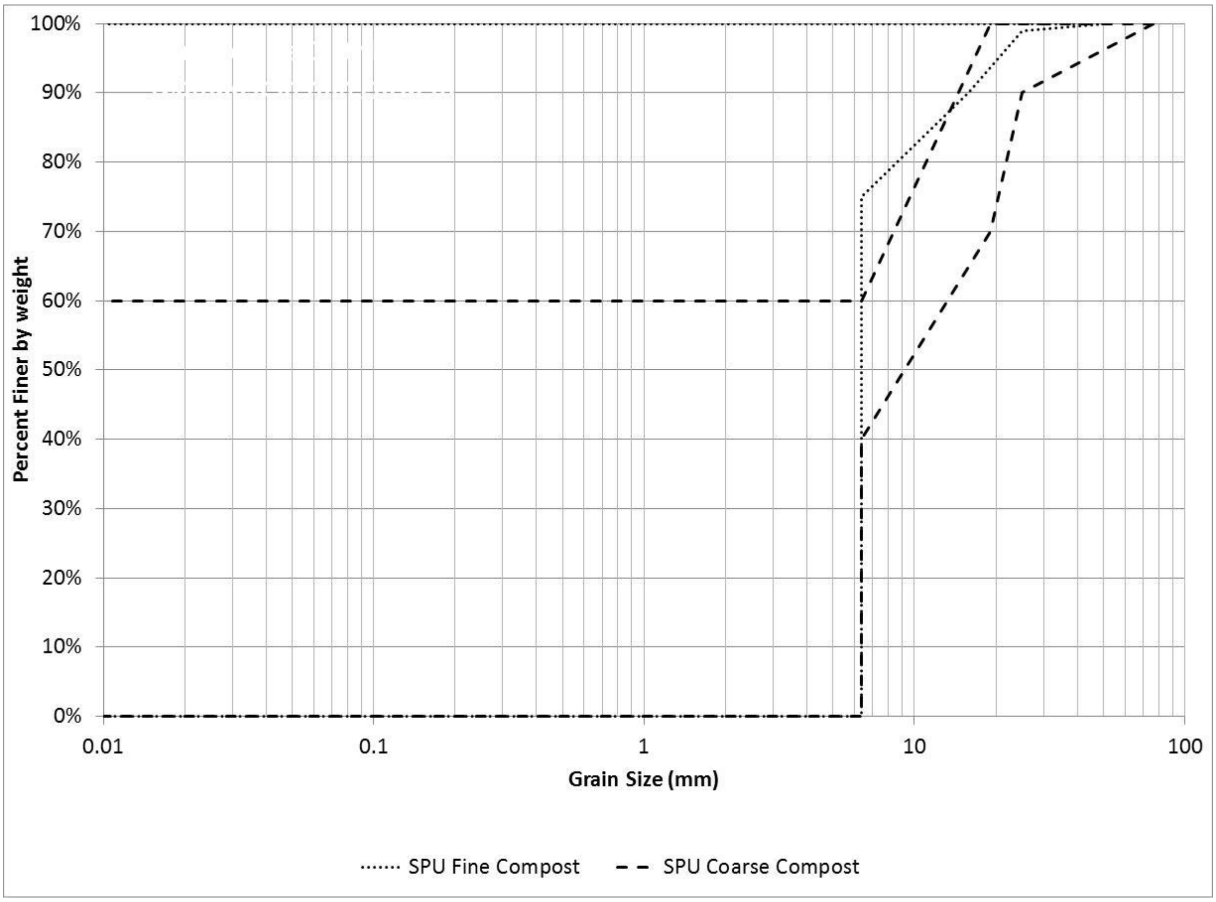


Figure 6 Compost PSD limits for candidate bioretention filter media according to Seattle Public Utilities (2008).



2.6 Compaction during construction

Compaction can determine whether or not a bioretention cell will have acceptable hydraulic performance, where media are vulnerable to compaction. Table 5 shows a selection of installation guidelines for bioretention filter media.

Table 5 Recommended installation methods

Jurisdiction	Guideline on lifts	Guideline on compaction
Prince George's County, (2007)	200 to 300 mm lifts	Natural compaction with light watering
ARC TP10 (2003)	300 to 400 mm lifts	Loose compaction by light tamping with backhoe bucket
North Shore City Council (2009)	300 mm lifts	Natural compaction with wetting of soil
Melbourne (FAWB 2009b)	Two lifts if depth is over 500 mm	Light compaction; single pass with vibrating plate for small systems; single pass with roller for large systems
Seattle Public Utilities (2008)	Loose lifts	Compact to 85 to 90% of modified maximum dry density
California Stormwater (CASQA 2003)	460 mm or greater lifts	Light compaction

There is clearly no consensus on the optimum level of compaction required for bioretention. Arguably more worrying is the lack of guidance on what 'light compaction' and 'compact to xx% of modified maximum dry density' mean, and how it is achieved in practice. In the construction industry, a specification of "compact to xx% of maximum dry density" is achieved through any sort of means available to the contractor. The contractor is understood to have the knowledge and experience to know how much compaction to perform, and if the accuracy of compaction is required to be tested, a cone or penetrometer test may be performed. As the specified compaction is relative to the dry density of the material, it is therefore assumed the contractor has expertise and experience with the material.

2.7 Pollutants

Pollutants that can be found in stormwater runoff include pathogenic bacteria, sediments, phosphorus, nitrogen, zinc, copper, lead, and oil and grease, among others (USEPA et al. 2004). It is well established that the amount of these pollutants found in stormwater runoff substantially increases with urban development. In Auckland, heavy metals in stormwater runoff, particularly zinc and copper (Timperley, Williamson, & Horne, 2005), have been identified as primary pollutants of concern, contributing to adverse effects on the social, cultural, and economic values of the region (Boston Consulting Group 2004). The Hauraki Gulf Forum's 2011 State of our Gulf report indicates that nutrients (particularly nitrogen) are causing measureable impacts within that receiving environment. Several Auckland Council and Auckland Regional Council technical reports discuss stormwater pollutant sources and impacts, so the information will not be repeated here.

International research identifies key pollutants that bioretention may effectively mitigate include sediment, heavy metals and nutrients. Relatively little information has been identified that clearly links media chemistry to water quality performance, although it is currently a topic of interest within the international community, with research occurring at several universities. To date, the only restrictions for bioretention media chemistry related to water quality have been identified from North Carolina (USA). Media is required by the state governing body to show a phosphorus "P index" of 10 - 30 if it is to be used as bioretention media[2]; installations are field tested for compliance. Table 6 relates media depth to pollutant removal according to international research.

[2] In NC, the use of the P-index originated from agricultural soil assessments to give a general indication of levels of available phosphorus for either supporting plant growth or causing nutrient-enrichment problems for downstream receiving environments. It is relevant in a bioretention application for an ability to ab/adsorb phosphorus from stormwater runoff. Further information on NC's P-index is available from: http://www.sera17.ext.vt.edu/Documents/P_Index_for_%20Risk_Assessment.pdf and http://www.ncagr.gov/agronomi/obpart1.htm#irs

Table 6 Required media depths for specific pollutant removal

Pollutant	Source	Depth
TSS	DiBlasi et al. (2009)	300 mm
Metals	Li & Davis (2008a), Hatt et al. (2009)	300 mm
Oil and grease	DiBlasi et al. (2009)	300 mm
Phosphorus	Hsieh et al. (2007), Passeport et al. (2009), Hatt et al. (2009)	600 mm (min) 900 mm (preferred)
Nitrogen	Passeport et al. (2009)	900 mm
Temperature	Jones & Hunt (2009)	900 mm (min) 1,200 mm (preferred)

Most heavy metals in urban stormwater runoff come attached to suspended solids (Bodo 1989). The concentrations attached tend to increase with decreasing particle size (Liebens 2001), due to the larger surface area in finer sediment being able to hold more pollutant ions (Dong et al. 1984). Zanders (2005) found vehicle-derived sediments in New Zealand were mostly material finer than 250 ∝m, and these small particles contained markedly higher heavy metal concentrations than larger particle sizes. Finer sediment particles are more easily transported in stormwater, making the suspended solids an important factor to treat to reduce heavy metals. Sediment and particulate-attached heavy metals removal by filtration mechanisms provided by bioretention are relatively well understood in the literature.

Despite the strong link between heavy metals and suspended solids, one of the most important indicators of bioavailability of heavy metals reaching ecosystems lies in analysing the dissolved portion of heavy metals (Herngren et al. 2005). Dissolved heavy metals cannot be treated by filtration, but require adsorption, precipitation, or other chemical reaction for removal.

Bioretention cells incorporate several pollutant removal mechanisms to treat stormwater (Table 7 and Figure 2). Non-specific adsorption (often simply called adsorption) is chemically considered as weak, electrostatic bonding. The process of adsorption happens rapidly. Soil particles only contain a limited number of available adsorption sites.

Potential for cation adsorption is measured by cation-exchange capacity (CEC). A CEC value is the maximum quantity of cations a soil particle can hold on a negatively charged site, at a constant pH. A higher CEC value indicates adsorption of pollutants is more likely to occur.

Base saturation is a relative measure of how many exchange sites are actually available for a reaction to take place, and is expressed as a percentage of CEC.

Specific adsorption of heavy metals (including Cu^{2+} and Zn^{2+}) and phosphate ions (PO_4^{3-}) can occur if specific types of minerals are present. Specific adsorption refers to the formation of a stable complex between ions (in this case, pollutant ions), and particular functional groups at the surface of a soil particle. Specific adsorption is less rapid than the electrostatic adsorption (Lucas & Greenway 2011). The most notable minerals for specific phosphate adsorption contain iron hydroxide and/or aluminium hydroxide groups which can undergo ligand exchange reactions with phosphate ions. For heavy metals, the principal minerals for specific adsorption are the same as for phosphate, as well as manganese hydroxide (McLaren & Cameron 1996).

Table 7 Pollutant removal mechanisms in bioretention (Hunt et al. 2012)

Pollutant removal mechanism	Pollutants
Adsorption to soil particles Plant uptake	Dissolved metals and soluble phosphorus Small amounts of nutrients (phosphorus and nitrogen)
Sedimentation Filtration	Total suspended solids, floating debris, soil-bound phosphorus, soil-bound pathogens
Microbial processes	Organics, pathogens
Exposure to sunlight and dryness	Pathogens

3.0 Literature review: media characteristics vs. performance

3.1 Bioretention media laboratory tests

Laboratory tests have been widely conducted to determine the capability of bioretention cells for water quality improvement. Typically, columns are filled with different materials and the leachate tested to determine the effectiveness of removal. A combination of the materials should be used to find the optimal mix to remove various pollutants and provide the most effective control for stormwater quality.

Laboratory tests often use synthetic stormwater to replicate real world conditions. Pitt et al. (2011) found the inherent chemistry of stormwater to be substantially different to artificial mixes found in literature, and therefore used modified stormwater to test media, even in laboratory conditions. Stormwater was modified by collecting daily runoff and increasing the concentration of several pollutants to reach the 90th percentile of target concentrations.

Davis et al. (2003) found over 95% removal of copper, lead and zinc with various changes in test conditions such as flow duration, intensity, stormwater pH, and influent pollutant concentrations. Phosphorus, Total Kjeldahl Nitrogen (TKN) and ammonium removal levels varied from 50% to 80%, increasing with the depth of media. Nitrate removal was less than 20% or in some cases nitrification was present and nitrate was produced. Nitrate reduction was also affected by varying flow intensity and duration.

Bratieres et al. (2008) performed large scale column tests to optimize various factors for nutrient and sediment removal using bioretention. 125 columns of varying plant species, filter media, filter depth, filter area and pollutant inflow concentration were tested. The different filter media used were sandy loam, sandy loam with addition of 10% vermiculite and 10% perlite, and sandy loam with addition of 10% leaf-compost and 10% mulch (all percentages by volume). These factors were then replicated with three differing media depths of 300, 500 and 700 mm. Plants were watered as required for six months to allow establishment.

At the beginning of the trials the sandy loam had a mean hydraulic conductivity of 186 mm hr^{-1}, which eventually decreased to 88 mm hr^{-1} after seven months of twice-weekly dosing of 25 L of “semi-natural” stormwater. Semi-natural stormwater is water collected from a stormwater pond inlet which is sieved to achieve a targeted TSS concentration. Pollutants measured were TSS, nitrogen and phosphorus. Results showed all configurations of biofilters were consistently effective at TSS removal (over 95% reduction). TSS removal had little change over time and was not affected by clogging in the filter media itself. These findings are consistent with previous studies (Hatt et al. 2006; Hatt et al. 2007; Hseih & Davis 2005a; Hseih & Davis 2005b). For nitrogen removal, the soil media without additional organic matter proved to be significantly more effective after the initial stage of testing due to media with additional organic matter slowly leaching nitrogen as organic matter broke down. Similar results occurred with phosphorus removal when the breakdown of organic matter released phosphate.

With increasing filter media depth there was a net production of nitrogen oxides, but no difference in ammonia and organic nitrogen. The reason for the difference is attributed to the different root systems of vegetation when they grow under different depths of soil. Different media depths cause differing root growth rates and structures which in turn cause different nitrogen uptake capacities and preferential flow paths. Filter media depth did not have any effect on phosphorus removal. The processes involved with phosphorus removal were recognized as filtration for particulate phosphorus, and plant and microbial uptake and natural sorption for phosphate.

Bratieres et al. (2008) concluded maximum performance would be achieved by planting bioretention cells with particular plant species (*Carex appressa* or *Melaleuca ericifolia*) for nutrient removal, and be made of sandy loam filter media without additional organic matter.

Hatt et al. (2008) conducted a laboratory scale study to provide an overall assessment of hydraulic and pollutant removal behaviour of sand and soil based filters. Emphasis was given to the influences of time, cumulative inflow sediment, cumulative water volume, wetting and drying, and compaction on hydraulic conductivity. Pollutants tested were TSS, nitrogen, phosphorus, copper, lead and zinc.

Six different filter media types were tested. One was sand based (fine sand) and the other five soil based (sandy loam, sandy loam with a 20% synthetic soil ameliorant commercially available in Australia), sandy loam with 10% vermiculite and 10% perlite, sandy loam with 10% compost and 10% mulch, and sandy loam with 20% compost and 20% mulch on charcoal drainage layer, with all percentages by volume).

To get the test columns to a “mature” stage, the filters were flushed with clean water daily for 17 weeks. Semi-synthetic stormwater was produced based upon typical target pollutant concentrations from dense Australian urban catchments; pollutants included TSS, nitrogen, phosphorus, copper, lead and zinc. In order to simulate a wet period, filters were dosed three times per week with an equivalent of a six month average recurrence interval storm for a filter sized at 2% of the effective impervious catchment area. After 42 weeks of wetting and drying, the hydraulic conductivity of soil based media reduced as much as 68% (from 2,329 mm hr^{-1} to 749 mm hr^{-1}) while the sand based media reduced by less than 3% (from 260 mm hr^{-1} to 254 mm hr^{-1}). The high reduction in the soil based media is attributed to compaction as witnessed by their decreased media depths (from 1 m down to 0.72 m in the most extreme case). Infiltration rates were also affected by clogging of captured sediment near the filter surface.

All six different media types showed over 90% reduction of TSS, copper, lead and zinc over 42 weeks. On the other hand only the sand based media removed phosphorus and nitrogen, while all soil based media leached phosphorus and nitrogen. All pollutants were found to have been most effectively trapped in the top 20% of the filter profile (0.2 m), and the authors recommend scraping off the top 2 to 5 cm of the filter surface every two years to prevent hydraulic failure as well as remove build-up of pollutants.

Barret et al. (2011) tested concrete sand (specified in ASTM C-33), masonry sand (ASTM C-144), and a concrete sand with topsoil mix (Table 8).

Table 8 Characteristics of media presented in Barret et al. 2011

Media	Sand (%)	Silt (%)	Clay (%)	Organic Matter (%)	CEC (meq/100g)
Concrete sand	88	10	2	0.1	5.3
Concrete sand with topsoil mix	73	18	9	0.4	9.8
Masonry sand	94	2	4	0.1	0.9

Pollutants tested were TSS, nitrogen and phosphorus. The concrete sand was the poorest performing for removal of all three pollutants. The topsoil mix had higher TSS and total nitrogen removal rates than masonry sand. Total phosphorus removal was similar between the topsoil mix and masonry sand, both performing better than concrete sand. The finer topsoil material was identified as the reason for the better TSS removal rate.

Both concrete sand and masonry sand only have 0.1% organic matter by weight, but no additional silt, clay or organic matter was required to support vegetation. The effect of vegetation on pollutant removal was also studied (Table 9). The presence of vegetation had no effect on TSS removal. However, vegetation did improve total nitrogen, total phosphorus and dissolved phosphorus removal.

Table 9 Effect of vegetation for pollutant removal presented in Barret et al. (2011)

	Pollutant removal			
	TSS	**Total nitrogen**	**Total phosphorus**	**Dissolved phosphorus**
Vegetated	88 - 97%	59 - 79%	77 - 94%	71 - 94%
Non-vegetated	88 - 97%	18 - 25%	58 - 80%	43 - 75%

3.2 Bioretention media with additives

Various studies have used soil amendments to further increase the pollutant removal capability of bioretention filter media. The removal of particulate phosphorus is from a combination of sedimentation and inert media filtration. These mechanisms are ineffective at removing dissolved phosphorus; dissolved phosphorus requires the media to have an anion exchange (adsorption) capacity (AEC) in order to be removed from stormwater. Phosphorus adsorption increased with additives such as aluminium-based drinking water treatment residual, triple-shredded hardwood bark mulch, red mud (a by-product of bauxite processing and a strongly alkaline material high in aluminium oxides), and Krasnozem soil (highly aggregated red clay soil) (Lucas & Greenway, 2011; O'Neill & Davis, 2012 a, b). Several New Zealand source materials were identified in Section 2.1 that may contribute to phosphorus removal; however, a commercially-available, highly consistent resource has not been identified. Drinking water treatment plant residuals are receiving significant attention in stormwater filter media research in the USA, as they are a readily-available waste product from municipal plants.

Control of nitrogen increased with amendments including zeolite, sulphur and wood chips (Tarkalson & Ippolito 2010; Ergas et al. 2010). Similarly, metal ion sorption increased with addition of crab shell, or *Sargassum* (marine algae) (Vijayaraghavan et al. 2010). More unconventionally treated pollutants including dioxins, mercury, perchlorate, oil, grease and radioactive components were also shown to be treatable with bioretention containing virgin coconut-hull granular activated carbon (Pitt & Clark, 2011).

3.3 Field studies

In North Carolina field tests have repeatedly shown very low total phosphorus (< 0.15 mg L^{-1}) and total nitrogen (< 1.0 mg L^{-1}) in stormwater effluent from bioretention cells. The filter media used is 85% medium grain or coarser sand, 10% silt or clay (from specific sources) and 5% organic amendment (usually yard waste compost) by volume (Hunt et al. 2008). It is noted that North Carolina requires post-installation testing of filter media to ensure compliance with the local P-index limit (Section 2.7).

Further field studies in North Carolina found evapotranspiration and exfiltration accounted for 42% and 31% of inflow runoff for 0.9 and 0.6 m deep filter media respectively. Estimated annual pollutant load reduction for nitrogen, phosphorus, and TSS were 19, 44, 82% for the 0.9 m deep cell, and 21, 10, 71% for the 0.6 m deep cell respectively (Brown & Hunt, 2011a). The primary reason for pollutant load reduction was due to the significant runoff volume reduction.

Field monitoring of 28 rain events over 2007 to 2008 of a bioretention cell in Blacksburg, Virginia USA found reduced flow volumes of 97% and reduced peak flows of 99% (DeBusk and Wynn, 2011). Cumulative mass reduction of sediment, total nitrogen and total phosphorus reflect volume control; all three measures were reduced by over 99% by mass. The bioretention cell fill media was 88% washed medium sand (0.2 mm to 0.63 mm), 8% clay and silt, and 4% leaf compost by volume. No compaction was implemented, and the cell was overfilled to allow for settling. It should be noted that the bioretention cell was much deeper (1.8 m) than what is deemed standard (0.6 to 1.2 m), and the systems were installed over limestone. These factors may have influenced the impressive results.

Two bioretention sites built and monitored in Michigan indicated nutrient removal was primarily based on volume reduction and not concentration reduction (Carpenter & Hallam 2010). The two bioretention cells had contrasting filter media. The first was 20% compost, 50% sand, and 30% topsoil filter media; the second 80% compost with 20% sand. Both bioretention cells leached nutrients when assessed on a concentration basis, but removed nutrients when assessed on a mass basis (Table 10). For comparison, the 20/50/30 cell generally retained a larger percentage of inflow and stored more water than the 80/20 cell. The effect of the volume reduction is evident in the pollutant mass removal for nutrients, especially nitrogen.

Table 10 Field study by Carpenter & Hallam (2010)

	Pollutant removal		
	TSS	**Total phosphorus**	**Total nitrogen**
20/50/30 cell	79.3%	97.2%	90.8%
80/20 cell	97.9%	76.9%	19.9%

Monitoring of two bioretention cells in Kansas showed removal of suspended solids and zinc concentrations, but increased nitrogen and phosphorus concentrations. The bioretention media was a soil mixture containing 50% hardwood compost and 50% sand by volume. It is hypothesized the increased nitrogen and phosphorus is a result of the breakdown of the organic material (i.e. compost) (Peltier & Carbone, 2011).

One of the major concerns when designing a bioretention cell is the potential for clogging to occur at the surface of the cell from fine particles washed in with stormwater (at least until plants develop) or crusting (or sealing) from breakdown and sorting of substrate particles at the surface. Both clogging and sealing reduce infiltration rates into the fill media, and hence reduce the effectiveness of the device. A study of a bioretention cell in Villanova, Pennsylvania, found that over a seven year period, where fines have accumulated in the cell, there was no significant change in infiltration rates (Jenkins et al. 2010). Conversely, Brown & Hunt (2011b) found severely undersized bioretention cells in North Carolina clogged in less than one year, and after removal of the clogging layer the infiltration rate increased by almost a factor of ten. Clogging was attributed to granite fines washed from the gravel base layer of an in-construction asphalt parking lot. Li & Davis (2008a) found clogging to reduce the seepage rate of laboratory columns by an approximate factor of ten after simulating a year’s worth of storm water loading.

Li & Davis (2008b) found no such clogging after 1.5 years during field monitoring. The vegetation and fauna, such as earthworms, worked to open and loosen up the media, thus maintaining an acceptable permeability (providing confirmation of laboratory studies in section 0). Column tests that lack vegetation and fauna probably overestimate the state of clogging in bioretention cells in the medium term, and may be reflective of the maximum potential for clogging. Certainly the ability of a bioretention cell to remove sediment from stormwater runoff is one of its design intents; proper design and installation including consideration of construction timing and long-term catchment loading potential is essential for success.

3.4 Summary

Nutrient removal or release by bioretention is heavily influenced by media composition. Higher organic content media tends to leach nitrogen and phosphorus, but may be mitigated by various soil amendments. Heavy metals and TSS seem well controlled by bioretention regardless of the media composition. Laboratory studies greatly simplify field systems. Factors such as presence or absence of vegetation, earthworms and other organisms, weather patterns, and runoff composition create confounding or mitigating affects. Control of runoff volume by bioretention is a substantial contributor to overall pollutant removal effectiveness, regardless of changes to actual concentrations. In other words, bioretention usually beneficially reduces runoff mass even if certain pollutant concentrations increase.

4.0 Methodology

4.1 Visiting suppliers and gathering material information

Most sand and organic materials' suppliers were visited in March and April 2011 to survey the types of material available locally. Material was obtained from Winstone Aggregates in June and July 2011. Specific information from suppliers about the possible materials included:

- Material type and source(e.g. pumice, river sand, beach sand, scoria)
- Specific particle sizes, grading, weight
- Washed or unwashed (determines content of fine material)
- Any particular chemical properties (e.g. high iron content, high cation exchange capacity [CEC], etc.)
- Product quality control
- Availability of supply, consistency of supply
- Typical uses/applications (other than bioretention)
- Price
- Other supplier/industry insight

4.2 Particle size distribution of individual components

To narrow down the range of suitable aggregates for bioretention filter media, PSD was used as a primary screening process. The ability of local materials to meet gradation limits summarized in Section 2.5 and Figures 4-6 were investigated.

PSD tests were carried out using a dry sieve analysis as per ASTM C136-06: Standard Test Method for Sieve Analysis of Fine and Coarse Aggregates (ASTM International, 2011a). The "local" equivalent test method is AS1289.3.6.1-1995: Determination of the particle size distribution of a soil – Standard method of analysis by sieving (Australian Standard, 2009), and is no different to the ASTM Standard.

4.3 Compaction testing

Compaction testing was completed to investigate characteristic differences between soil based and sand based bioretention media. Two commercially available bioretention mixes (CM1 and CM2), four sands (East Coast Sand, Woodhill Black Sand, Pumice Sand and No.3 Sand), as well as Pumice Sand + 10% v/v compost mix, were subjected to compaction testing using ASTM D1557-09: Standard Test Methods for Laboratory Compaction Characteristics of Soil Using Modified Effort (2,700 kN-mm^3) (ASTM International, 2011b) (Figure 7). The standard uses repeat tests to assess the effects of moisture content and compaction on bulk density. The result of testing is a curve which describes the change in maximum densities after compaction according to a range of initial media moisture contents.

Figure 7 Equipment used in modified proctor compaction test (as per ASTM D1557-09 (left); Jack system to remove compacted soil sample from proctor (right)

4.4 Laboratory replication of field compaction

To replicate real compaction practices in the field for the small scale laboratory tests, it was necessary to determine the equivalent compaction procedure in the laboratory. Field compaction was replicated in the laboratory using bulk density data from a 2009 field study of bioretention cells along Corban Ave (Albany, North Shore) constructed in 2006 (Landcare Research, unpublished data). The construction of the bioretention cells had used CM1 and presumably followed TP10 guidelines for compaction (light tamping with backhoe bucket). The level of compaction was replicated by setting the desired density by controlling volume and mass of CM1 in a column, and measuring the amount of blows with a modified proctor hammer (with a force of 2,700 kN-m/m^3) it took to achieve compaction of required mass into the required volume (hence achieving the required density).

4.5 Hydraulic conductivity testing

Hydraulic conductivity assesses the ability of a bioretention filter media to meet the surface infiltration/drawdown objectives and pollutant removal in the field. FAWB recommends testing hydraulic conductivity using the ASTM F1815-06: Standard Test Methods for

Saturated Hydraulic Conductivity, Water Retention, Porosity, and Bulk Density of Putting Green and Sports Turf Root Zones (ASTM International, 2011c) method, whereas WSU recommends ASTM D2434-68: Standard Test Method for Permeability of Granular Soils (Constant Head) (ASTM International, 2011d). SPU and others have developed a modification to ASTM 2434, including correction factors, since it is a granular soil test that needs special attention for soils with significant OM content (Hinman, personal communication 2011), but it is not yet publically available. These standard methods involve miniature permeameters and are extremely small scale tests. For the objectives of this research, a larger set-up (Figure 8) was preferred to more closely mimic a bioretention system (including the construction and compaction phase) used in the field.

The test set up was intentionally simple in an attempt to limit the scope of the project and focus on the filter media itself. A mulch layer, transition layer and drainage layer were removed to decrease the factors affecting the final results, leaving only a soil fill layer, hence simplifying analysis.

1500 mm columns of transparent Perspex pipe with a 140 mm inner diameter were used to replicate bioretention cells in the lab (Figure 8a). Water was applied through individual pipes and valves for each bioretention column (Figure 8b). Media was supported by a 1 mm mesh at the bottom of the column (acting as a drainage layer) which allowed water to flow through, and limited the media from escaping (Figure 8c). The depth of ponding on top of the media layer was fixed by drilling a hole in the pipe at the desired ponding level (300 mm above media), which allowed excess water to spill out of the column when the ponding level was reached (Figure 8d).

Based on literature review of worldwide bioretention cell guidelines (Section 2), bioretention media depths seem to be decreasing and ponding depths increasing. Following these trends, a relatively shallow media depth of 600 mm and relatively deep ponding depth of 300 mm were chosen for testing.

Homogeneous mixing of sand and compost was achieved using a cement mixer, more specifically a “creteangle multi-flow mixer” (Figure 9). Initially, proportions were measured by weight. However, for ease of comparison with international guidelines and literature, and in recognition of likely blending processes, the laboratory process was changed to proportioning by volume. The final procedure was to measure the correct volumetric proportions in the mixer and mix for 30 seconds at the “as delivered” moisture. Mixes were tested with arbitrary high or low additions of compost.

Mixed media was placed into the columns in 300 mm lifts. Compaction was either by wetting each layer or light tamping. "Compaction by wetting" is the term adopted herein to refer to compaction induced by adding water (no mechanical action) to promote settling. For each 300mm lift, water was applied from the top of the column to a condition when it ponded for an extended period of time, and effluent was visually estimated to be constant. It was assumed that saturation was reached at this point.

Figure 8a-d Clockwise from top left: Laboratory columns; water inflow system; mesh attached to bottom of column; constant ponding depth

Proportion by volume to proportion by weight can be converted using bulk densities (Table 11) and equation 1:

$$W_c = \frac{v_c \times \rho_c}{(v_c \times \rho_c) + (v_a \times \rho_a)} \qquad \text{Equation 1}$$

Where

W_c = % weight of compost

v_c = % volume of compost

ρ_c = bulk density of compost

v_a = % volume of aggregate

ρ_c = bulk density of aggregate

Table 11 Bulk density of material[1]

Material	Abbreviation	Average Bulk Density[2] (g/cm^3)
Commercial Mix 1	CM1	0.685
Commercial Mix 2	CM2	0.641
Green-waste Compost	Compost A	0.749
Bark based Compost	Compost B	0.532
No.3 Sand		0.925
East Coast Sand	ECS	0.972
Woodhill Black Sand	WBS	0.996
Pumice Sand	PS	0.505

1. Bulk densities determined as freely settled in standard volume core, no compaction, as delivered, at delivered water content
2. Average of three replicate measurements.

Figure 9 Cement mixer used to mix homogeneous bioretention filter media mix

The hydraulic conductivity test on the columns was a falling head permeability test from the maximum ponding depth (300 mm above media surface). Before each test run, water was applied from the top of the the column until flow conditions suggested saturation was achieved. Extended ponding and estimated constant effluent flow rate was used to judge that saturation had been reached. The time taken for water to fall from the maximum ponding depth to the media surface was measured and saturated hydraulic conductivity calculated in mm hr^{-1}. Ten test runs were performed for each different media column. Replication (duplicate testing) was performed on two columns for quality assurance.

The influence of design parameters on hydraulic conductivity were tested by assessing six different variables. The variables tested were:

- Media depth and ponding depth
- Type of proprietary mix
- Type of aggregate material
- Type of organic material
- Proportion of organic material
- Level of compaction

Thirteen columns tested six different variables (Table 12).

Table 12 Media composition (% v/v) subject to hydraulic testing in laboratory columns[1, 2]

Media	Compaction Method Applied to 300 mm Lifts	Variable Tested
CM1 (total 14.36% organic matter)	Wetting and settling	TP10 (2003) media depth 1 m and ponding depth 220 mm)
CM1 (total 14.36% organic matter)	Wetting and settling	Commercially available mix
CM2 (total 16.94% organic matter)	Wetting and settling	Commercially available mix
CM1 [Lightly Tamped] (total 14.36% organic matter)	Wetting and settling	Compaction
CM2 (total 16.94% organic matter)	Lightly tamped[3]	Compaction

Media	Compaction Method Applied to 300 mm Lifts	Variable Tested
94% No.3 Sand (ALS) + 6% Compost A	Wetting and settling	% Compost
88% No.3 Sand (ALS) + 12% Compost A	Wetting and settling	% Compost Compost type
84% No.3 Sand (ALS) + 16% Compost B	Wetting and settling	% Compost Compost type
57% No.3 Sand (ALS) + 43% Compost B	Wetting and settling	% Compost
90% No.3 Sand (ALS) + 10% Compost B	Lightly tamped	Compaction
90% East Coast Sand + 10% Compost B (total 0.88% organic matter)	Wetting and settling	Aggregate type
90% Woodhill Black Sand + 10% Compost B (total 0.46% organic matter)	Wetting and settling	Aggregate type
90% Pumice Sand + 10% Compost B (total 1.81% organic matter)	Wetting and settling	Aggregate type
90% Pumice Sand + 10% Compost B (total 1.81% organic matter)	Lightly tamped	Compaction

1. All mixes were homogeneous and mixed via method described in section 4.5. Columns were of media depth 600 mm and ponding depth 300 mm, except the first CM1 listed in the table.
2. Duplicate testing on CM1 and CM2.
3. Light tamping, equivalent to TP10 (ARC 2003) was determined to be 15 blows with a modified proctor hammer. Refer to section 5.4.

4.6 Chemical analysis of material

Chemical analysis of individual material components were performed by Landcare Research. Properties of interest were identified as they related to plant germination and/or growth potential, including: pH, organic carbon, total nitrogen, carbon:nitrogen ratio, Olsen phosphorus, total phosphorus, AEC, CEC, and base CEC saturation. Chemical tests were carried out by Landcare Research on the < 2 mm fraction (samples are sieved, then ground) and results were reported on a dry mass basis. Chemical activity is dominated by finer particles (clay and silt). The majority of soil chemistry test methods are after Blakemore et al. (1987), which are briefly described on the Landcare Research website: http://www.landcareresearch.co.nz/services/laboratories/eclab/eclabmethods_soils.asp. Methods for testing Total Carbon and Total Nitrogen are based on Leco (Laboratory Equipment Corporation, undated), also briefly described on the Landcare Research website. Total organic matter was determined from Total Carbon by multiplying by 1.72 (Leco 2003; Metson et al. 1979; Miller undated; Nelson and Sommers 1996).

4.7 Water quality testing

4.7.1 Media preparation and column setup

Five distinct media mixes were chosen to undergo water quality testing based on their performance following the hydraulic testing stage (Table 13). Laboratory scale 140 mm (inner diameter) bioretention columns with the five chosen filter media were built for the sole purpose of water quality testing.

Prior to the first dosing, columns were flushed with clean tap water to remove “first flush” contaminants. Tap water was tested for zinc and copper content, both of which were below detection limits. For the sand-based mixes, water used for compaction by wetting procedure was assumed to flush contaminants. It took approximately 4 hr for water to flush through the system during this process. For the commercial mixes, the visual cue of an improvement in effluent colour signalled the end of flushing. In a field application, is it impractical to wash media prior to installation. However, pollutants emitted from the media in the first flush are likely to be insignificant compared to the cumulative pollutant load removal over the device’s lifetime.

Table 13 Filter media chosen for water quality testing

Abbreviated name	Media[1] (% v/v)	Compaction	Hydraulic Conductivity (mm hr^{-1})
ECS+	90% East Coast Sand + 10% Compost B	Wetting and natural settling every 300 mm	80
WBS+	90% Woodhill Black Sand + 10% Compost B	Wetting and natural settling every 300 mm	400
PS+	90% Pumice Sand + 10% Compost B	Wetting and natural settling every 300 mm	340
CM1	CM1	Light tamping[2] every 300 mm	180
CM2	CM2	Light tamping every 300 mm[2]	240

1. Each mix was tested in duplicate.
2. Light tamping, equivalent to TP10 (ARC 2003) was determined to be 15 blows with a modified proctor hammer. Details in Sectio n 5.4.

Each test run involved dosing all columns with 20 L (see section 4.7.3.1) of synthetic stormwater. Influent was delivered by a 10 L bucket positioned above each individual column (Figure 10a), with the flow rate controlled to keep a constant 300 mm ponding depth until buckets were almost empty. This procedure was completed twice for each test run to make a total of 20 L per column. Effluent was collected in individual buckets at the bottom of each column (Figure 10b).

Figure 10a-b Influent dosing buckets and effluent collection buckets

4.7.2 Dosing schedule

A dosing schedule (Table 14) was developed to obtain a balance between two objectives:

- Estimating storm event-based pollutant removal efficiency of the media
- Estimating comparative total pollutant removal potential and useful life amongst the different media

The initial six WQV doses (each from the 28.3 mm "water quality storm") were used to determine short term (event-based) pollutant removal efficiencies of the filter media when the cells are initially built. WQV doses are based on realistic stormwater pollutant concentrations, and therefore should produce results somewhat representative of field conditions.

Concentrated doses (5 year and 1 year) were then used to "age" the filter media by simulating multiple years' worth of pollutant loading in a short period of time (Lucas and Greenway 2008, 2011). The purpose was to provide a comparative estimation of total pollutant removal potential amongst the different media, and attempt to provide a rough estimate the life of the media (determined by the occurrence of a pollutant breakthrough). Subsequent WQV doses between the concentrated doses were used to estimate efficiency performance for media when aged 5 years, 10 years, and 15 years.

Table 14 Dosing schedule

Dosing concentration	Dosing ID
WQV	WQV1
WQV	WQV2
WQV	WQV3
WQV	WQV4
WQV	WQV5
WQV	WQV6
5 Year Concentrated	5YCONC1[1]
WQV	WQV7
WQV	WQV8
WQV	WQV9

Dosing concentration	Dosing ID
1 Year Concentrated	1YCONC1
1 Year Concentrated	1YCONC2
1 Year Concentrated	1YCONC3
1 Year Concentrated	1YCONC4
1 Year Concentrated	1YCONC5
WQV	WQV10
WQV	WQV11
WQV	WQV12
1 Year Concentrated	1YCONC6
1 Year Concentrated	1YCONC7
1 Year Concentrated	1YCONC8
1 Year Concentrated	1YCONC9
1 Year Concentrated	1YCONC10
WQV	WQV13
WQV	WQV14
WQV	WQV15

* # years of runoff using a concentrated dose of synthetic stormwater

4.7.3 Dosing procedure

4.7.3.1 Dosing volume

The dosing volume was established based on calculations from Auckland Regional Council's TP10 (2003) Chapter 7 relating the water quality volume (WQV) of the catchment to the surface area of a field bioretention cell. The calculation was performed using dimensions of the laboratory columns, which are assumed to be a scale representation of a field scale bioretention cell. For the columns, the WQV was determined to be 20 L. Full calculations are presented in the Appendix.

4.7.3.2 Synthetic stormwater concentration

Three contaminants were studied: dissolved zinc, dissolved copper, and soluble reactive phosphorus. Zinc and copper were selected as they are known stormwater contaminants of concern in the Auckland region. Phosphorus was selected as it is often a pollutant of concern in fresh-water receiving environments (which are limited in the Auckland region). Nitrogen was not tested because of resource and logistical limitations (primarily not exceeding allowable holding times prior to sample analysis). Only dissolved contaminants were studied. Dissolved contaminants are generally bioavailable, leading to impacts for aquatic species when contaminants exceed threshold levels. For example, ANZECC water quality guidelines for heavy metals are written in terms of dissolved fraction concentrations. The dissolved portion of contaminants is considerably more difficult to remove in stormwater treatment devices (particulates are relatively easily removed through sedimentation and filtration); hence working with only dissolved contaminants is operating at worst case scenario. Long-term sediment loading to bioretention cells will likely affect performance, as it may contribute to clogging (thus creating potential for bypass) and decreased hydraulic conductivity. Unfortunately, obtaining consistent or reproducible ranges of particle sizes where the size of concern is silt, as well as controlling the particulate pollutant concentrations, (which both require testing and adjusting) is problematic.

Three different concentrations of synthetic stormwater were used (Table 15). A “WQV” dose is the concentration expected in typical urban stormwater runoff during an isolated storm event. The WQV concentration is based on freeway data from the (USA) National Stormwater Quality Database v1.1 (NSQD) (Pitt et al. 2004). The NSQD was specifically chosen as the data source due to the extensiveness of stormwater data compiled (over 3,770 separate storm events prior to 2004). All data in the NSQD has undergone quality assurance and quality control, and is classified into categories with differing land uses. There are currently no local (New Zealand) stormwater sources that carry such an extensive library of stormwater quality data.

Freeway stormwater concentrations were chosen to test the filter media under rigorous contaminant loading. Stormwater runoff sampled from freeways was significantly more polluted (in terms of the three contaminants of interest) than other land-use categories in the NSQD: mixed freeways, open space, mixed open space, and residential runoff. Median concentration values were used based on 105 and 130 freeway observations for dissolved zinc and dissolved copper, respectively. Phosphorus concentrations applied are lower than the median values reported in the NSQD (based on 22 observations) due to differences in reporting concentrations as phosphate (PO_4) versus phosphorus (P).

“1 year concentrated” (1YCONC) and “5 year concentrated” (5YCONC) doses were also employed in order to simulate aging of the media (Table 14). The 1 and 5 year concentrated doses are 1 and 5 years’ worth of WQV doses each concentrated into a single dosing volume (refer to the Appendix for calculations). In a laboratory setting, soil sorption potential can be influenced by the influent concentration, where increasing the concentration increases sorption. Nonetheless, the concentrated dose method allows for a comparison of relative performance amongst the various media mixes.

Table 15 Dosing concentrations and pollutant sources for synthetic stormwater

		Concentration (mg L^{-1})		
	Source	**WQV**	**1 year concentrated**	**5 year concentrated**
Dissolved zinc	$ZnCl_2$	0.050	2.500	12.400
Dissolved copper	$CuSO_45H_2O$	0.010	0.500	2.480
Soluble reactive phosphorus[1]	K_2HPO_4[2]	0.065	3.226	16.000

1. Phosphorus reported as P throughout this report.
2. Standard Methods (American Public Health Association et al. 2011) recommends KH2PO4 for preparing phosphorus concentrations by dissolving salts. Di-potassium was used due to availability at the time.

The method to prepare influent samples involves dissolving salts and a dilution step. A highly concentrated sample of synthetic stormwater was first prepared (1 mL:1 L, highly concentrated dose to target concentration ratio). The highly concentrated sample was then diluted to reach the target dosing concentrations. 10 mL of concentrated sample was measured using a syringe, and diluted into 10 L of tap water to create the synthetic stormwater.

Mechanical stirring was introduced after the 5YCONC1 dose to ensure contaminant concentrations were consistent within each 10 L dose and between every 10 L dose. Trial tests showed 1 minute of stirring for 10 L of synthetic stormwater was sufficient to produce a consistent concentration across the 10 L.

4.7.4 Sample collection

All procedures followed Standard Methods for the Examination of Water and Wastewater methodology (American Public Health Association et al. 2005). Sample collection procedures include: preliminary treatment of samples (APHA 3030), sample collection and storage (APHA 1060), quality assurance and quality control (APHA 3020).

All influent doses were prepared in two 10 L buckets (section 4.7.2). All effluent from a single column was collected by placing 10 L buckets at the bottom of the column. Two buckets were required to collect the entire volume of effluent from each dose.

The two influent buckets contained equal volumes of stormwater, so taking an equal volume of sample from each influent bucket would give an event mean concentration. 25 mL of sample was extracted from each bucket using a disposable 12.5 mL syringe and mixed together to create a 50 mL sample. Samples were filtered 12.5 mL at a time into centrifuge tube (for sample storage until analysis), where the effluent sample would have been mixed during handling. The same procedure was taken to obtain event mean concentrations from the two effluent buckets.

Samples were filtered as they were taken (0.45 μm membrane filter) and mixed in a centrifuge tube. A 50 mL sample was collected for both copper and zinc analysis. These samples were preserved with nitric acid and stored at 4°C. Another 50 mL sample collected using the same method was sent to an external laboratory (Section 4.7.5) for phosphorus analysis. The same procedure was used for both influent and effluent samples.

Replicate samples were taken for 20% of total samples. The columns replicated were randomly chosen for each dosing. Replicate samples required collection of two extra samples in addition to the normal sample. Replication was done for both influent and effluent samples.

4.7.5 Pollutant analysis

Analysis of influent and effluent pollutant concentrations was completed using a combination of University of Auckland laboratory facilities and by Watercare Ltd., and Hill Laboratories. Use of different laboratories was dictated by budget, equipment availability and data precision. Both Watercare Ltd. and Hill Laboratories are accredited by International Accreditation New Zealand (IANZ), which represents New Zealand in the International Laboratory Accreditation Cooperation (ILAC). The accreditation is internationally recognised. All procedures followed Standard Methods for the Examination of Water and Wastewater methodology (American Public Health Association et al. 2005). Analysis procedures include:

Determination of method detection limits (MDL) (1030), and sample analysis (3110 and 3111). Table 16 presents the different methods used for analysis along with the method detection limits (MDL).

Table 16 APHA et al. (2005) laboratory analytical methods with corresponding MDLs

Dose	Zinc		Copper		Phosphorus	
	Method	MDL (mg L^{-1})	Method	MDL (mg L^{-1})	Method	MDL (mg L^{-1})
WQV (1 - 6)	3111[1]	0.01	3125 B[4]	0.0005	No Data	-
WQV (7 - 15)	3125 B[4]	0.001	3125 B[4]	0.0005	4500-P F[3]	0.005
1YCONC (influent)	3111	0.01	3111	0.02	4500-P F[3]	0.005
1YCONC (effluent)	3125 B[4]	0.001	3125 B[4]	0.0005	4500-P F[3]	0.005
5YCONC (influent)	3111	0.01	3111	0.02	4500-P F[3]	0.005
5YCONC (effluent)	3111	0.01	3113[2]	0.001	4500-P F[3]	0.005

1. Conducted in University of Auckland Environmental engineering laboratory.
2. Conducted in University of Auckland Environmental engineering laboratory.
3. Analysis performed by Watercare Ltd.
4. Analysis performed by Hill Laboratories.

4.7.6 Data analysis and quality assurance

The Shapiro-Wilk test was used to check for normality of data distribution. Results showed data significantly deviated from a normal distribution; therefore parametric tests were not used. The distribution of influent pollutant concentrations were checked to ensure the same median concentration across all 10 columns using an independent sample Kruskal-Wallis test. Effluent data from duplicate columns were checked using independent samples Mann-Whitney U test, to ensure the duplicates were not significantly different. Results from duplicate columns were then averaged before further data analysis and reporting.

To determine sampling precision, replicate samples were analysed and checked to ensure a relative standard deviation below 15%. Replicate sample results were averaged before further data analysis and reporting. To determine analyst accuracy of samples analysed at the university, three matrix spikes were conducted. Recovery of matrix spikes ranged from 97 to 107%.

4.8 Plant Growth Trials

Bioretention media developed in this research project have low organic levels, being ≤10% v/v compost, equating to 1 - 3% g/g total carbon. They also generally have very low clay and silt components, so generally have little ability to store and supply plant nutrients, and are considered to hold low volumes of water. The objective of the pot trial was to measure plant growth in the sand-based media to establish if they would support adequate plant growth in the short-term with low rates of slow-release fertilisation. A 'control' medium was used that had proven physical and chemical fertility for optimum plant growth. CM1 has established healthy plants over several years in Auckland, with high plant coverage and amenity (Fig. 11). This mix has 8 to 10% g/g total carbon (about 30% v/v).

Figure 11 Bioretention cells with commercial bioretention mix showing dense plant growth at about 12 months of age, 2009, Albany

Two plant species and two of the three sand-based media mixes were selected. *Carex secta*, green swamp tussock or purei, is a common wetland and bioretention plant that can grow to 2 m diameter and is often used in bioretention due to its rapid growth that quickly supresses weeds. It is highly tolerant of slow permeability but also tolerates some drought as it grows in areas with fluctuating water tables. In contrast, *Austrofestuca littoralis*, known as sand tussock or hinarepe, is highly tolerant of droughty conditions. Sand tussock grows to about 0.5 by 0.5 m and has an upright habit so is ideal for drier bioretention cells and edges bordering traffic or people. Both species are readily propagated and relatively inexpensive ($1.50 to $3 per root trainer). Pumice sand and East Coast Sand, both with 10% v/v Compost B, were selected for the growth trial. Woodhill Black Sand was not selected as it is very heavy to work with in large volumes by hand.

Twenty litres of each media was thoroughly mixed in a large container and used to fill eight 2.5 litre plastic pots (16 cm diameter). A single 'root trainer' seedling of the tussock-forming species *C. secta* or *A. littoralis* sourced from Taupo Native Plants Nursery was planted into each pot (i.e. four replicate pots of each species). No fertiliser was added to the commercial mix as testing showed organic and weak-acid extractable phosphate levels were high (>1200 and 400 mg kg^{-1} respectively), 5 to 10 times that of the sand-based mixes (140 to 360, and c.30 mg/kg respectively) (Section 5.6.2).

A low rate of 9 month slow-release fertiliser (N:P:K rating 13 : 5.7: 10.8 applied at 2.5 kg m^{-3}) was included in the sand-based mixes, equivalent to the recommended rate for growing trees.

Plants were grown in a shade house, rotated in position on an irregular basis, germinating plants noted, and plants watered as required. At the end of six months an 'average' plant from each treatment was photographed and all plants harvested. At this point plant roots were evident at the base of all pots. Foliage was removed 5 mm above the ground surface, weighed and basal circumference at 20 mm height measured. Foliage was dried in a forced-air oven at 80 C for 24 hours before reweighing. The ratio of wet to dry foliage was calculated as an indicator of green vs. dead leaf tissue; low ratios indicate a higher proportion of green leaves. The lower 200 mm section ('base') was weighed separately to generate an index of plant density; plants with a lower ratio have a denser base.

Tension tests were performed on limited mixes to quantify moisture held in the media as used by agronomists (Hillel 1971). Three pressure plate apparatus were used, depending on the tension being applied, as per standard practice (Gradwell and Birrell 1979). 150 mm diameter cores were used on low tension plates (1, 2.5, 5 and 10 kPa); 54 mm diameter cores on the Soil Moisture Corporation 'Catalogue 1500' pressure pot for 10 and 100 kPa (medium) tensions; and, 54 mm diameter cores on the Soil Moisture Corporation 'Catalogue 1600' were used for 300 and 1500 kPa (high) tensions. Oven-dry bulk density at a standardised level of compaction was assessed for lab-mixed samples. For comparison, hand-carved, intact cores removed from an Albany bioretention cell with CM1 were also tested.

5.0 Results: Materials' Supply, Physical Testing, and Component Chemistry

5.1 Visiting suppliers and gathering materials

The initial visit to landscape suppliers located around Auckland resulted in a range of materials which warranted further investigation (Table 17). The materials and/or products tested during the course of this research were largely selected on the basis of availability. For this reason, no natural soils were tested, although previous research has shown some soils are highly effective bioretention media. In any section, the information presented is not intended to endorse any particular product or company. Evaluations are limited to satisfying specific design objectives.

In 2011, two proprietary "rain garden" mixes were commercially available in New Zealand (Commercial Mix 1 and 2 [CM1 or CM2 in this report]). Chemistry analyses indicates both mixes have in excess of 30% v/v organic material, as total carbon levels are ~17% (CM1) and 14% (CM2) organic matter (Section 5.6). The supplier of CM1 indicated their mix includes 50% v/v organic material (composted bark and bark fines). High organic contents encourages plant growth by enhancing moisture storage and nutrient supply and storage, but may lead to physical degradation of the media and nutrient leaching, which is undesirable in a stormwater treatment system. Organic content for either commercially available product is substantially greater than specified in Table 1.

No.1 Sands are generally alluvial and consist of quartz and feldspar. No.3 from Auckland Landscape Supplies (ALS) was said to be pumice based. East Coast Sand is dredged marine sand with trace amounts of shell visible, confirmed by elevated Calcium levels (Section 5.6.2, Table 20a). There was concern over possible salt content, and the alkaline pH from the shell content. Salt and high pH (through decreasing available phosphate) may hinder plant growth. Woodhill Black Sand is dark and heavy due to its high iron content. It is excavated from the west coast of Auckland and is the cheapest sand sampled at an average of $55/m^3. Blocklayers sand is used by blocklayers in paving industry, as the name suggests. According to Central Landscapes, the sizing of Blocklayers sand is somewhere between the coarser No.1 sands and finer East Coast Sand. Winstone Pumice Sand ranges up to 2 mm diameter and is from Cameron Quarries in Otamarakau. SAP7/10 is scoria all passing 7 or 10 mm, and GAP7 is a general-uncategorized material all passing 7 mm. "All passing" refers to all material passing the specified sieve size.

Table 17 Available aggregates and composts considered as candidate materials, including prices from landscape suppliers around the Auckland region[1]

Supplier	Daltons	Central Landscapes	Auckland Landscape Supplies	Winstone Aggregates
Aggregates	No.1 Sand ($91/m^3$)	No.1 Sand ($82/m^3$)	No.3 Sand ($74/m^3$)	No.1 Sand ($69/m^3$)
	East Coast Sand ($91/m^3$)	East Coast Sand ($95/m^3$)	East Coast Sand ($97/m^3$)	No.3 Sand ($45/m^3$)
	Woodhill Black Sand ($50/m^3$)	Woodhill Black Sand ($57/m^3$)		Woodhill Black Sand ($46/m^3$)
		Blocklayers Sand ($82/m^3$)		GAP7[2] ($47/m^3$)
		SAP7 ($72/m^3$)		SAP7 or 10[3] ($40/m^3$)
				Pumice Sand ($51/m^3$)
Compost	Compost B ($95/m^3$)	Compost A ($108/m^3$)	Compost A ($76/m^3$)	
			Compost C ($89/m^3$)	

1 Prices excluding GST, as of March 2011

2 GAP7 refers to general-uncategorized material all passing 7 mm sieve

3 SAP7 or 10 refers to scoria all passing 7 or 10 mm sieve

The limiting factor when finding suitable composts are supply issues: namely consistency of supply and quality control of the supply (weed free, free of contaminants, meeting grading limits) while supplying in bulk quantities. Very few suppliers in Auckland can meet such standards, and hence only three compost products were sampled. Compost A is recycled green waste (garden waste such as tree branches and lawn clippings). Compost B is a blend of aged bark fines, untreated and aged sawdust, mushroom compost and chicken manure, with added Gypsum. Compost C is a 70%/30% (v/v) blend of peat/sand. The peat is from Ruakaka and has appreciable clay content.

Inconsistency in the marketing of materials across the different suppliers is apparent. For example, No.1 sand from supplier A could be similar to No.3 sand from supplier B. The numbering system is simply an in-house system characteristic of each supplier and is not designed to be a guide for the consumers to compare between different suppliers. The consequence of the inconsistent labelling system is each supplier's type of numbered sand had to be tested for suitability. Similarly, discussion with industry also revealed SAP7 (scoria all passing 7 mm, as marketed by retail suppliers) was in fact no different to SAP10 (scoria all passing 10 mm, as marketed by aggregate suppliers to the retail landscaping industry). PSD results in section 3.2 confirm the inconsistent sand labelling.

Since the source of GAP material is non-specific, properties other than maximum particle size are expected to be inconsistent. Discussions with concrete industry personnel who often use large quantities of the sand under investigation indicated that GAP7 is likely to consolidate into an impermeable layer when compacted, as well as being inconsistent in grading under 7 mm (due to being "all passing") (Crossland, personal communication 2010). This is an example of key information gained through dialogue with suppliers and industry, which ultimately saved time and resources that would have been put into testing an unsuitable material.

Information from one of the largest scoria suppliers in the industry revealed the scoria supply in the Auckland region is likely be unavailable in less than five years due to exhaustion of resource consented supply. Scoria was subjected to initial tests prior to this knowledge, and test results are still discussed in the remainder of this report for completeness. With the long term supply of scoria in doubt, it was ruled out of hydraulic conductivity and water quality testing. Full product information regarding a potential component is important, and this example further emphasises the importance of discussing the product with the supplier.

5.2 Particle size distribution

Eleven varieties of sands, three composts, and two commercially available bioretention mixes were sampled based on supplies available at four landscape suppliers throughout the Auckland region. PSD of each material is presented in Figure 12a-b, Figure 13a-b and Figure 14. Data were separated into multiple graphs to improve figure clarity. The uniformity coefficient (U) for each material was calculated using equation 2, based on PSD results.

$$U = \frac{D_{60}}{D_{10}} \qquad \text{Equation 2}$$

Where:

U = Uniformity coefficient

D_{60} = Grain diameter at 60% passing

D_{10} = Grain diameter at 10% passing

Figure 12a-b indicates that both Woodhill Black and East Coast Sands completely consist of fine material (100% passing 0.425 mm); therefore they both fall well outside the recommended range for bioretention media. Blocklayers Sand is also out of the range for over 50% of its composition and is poorly graded according to the ASTM D2487-10 classification system used by North Carolina (Hunt and Lord 2006). The remaining sands fall within or close to within the recommended PSD range. The remaining sands were similar "numbered" sands from four different suppliers, as well as Pumice Sand, GAP7 and SAP10. All numbered sands had similar U values of approximately 3, while East Coast and Woodhill Black Sand were more uniform with U values around 1 to 2. These are considered poorly graded per ASTM D2487-10. Pumice Sand, GAP7, and SAP10 are significantly less uniform with U values above 8, and are considered well graded. For comparison, the only guidance found for U was from Washington State which requires U $\geq$ 4 (Hinman, personal communication 2011).

All three composts fall within SPU's "fine compost" designation (Figure 14). SPU's compost limits are loosely interpreted, because the main emphasis of the limit is to minimize large grained particles in the compost. Large grains reduce surface area (and hence lower chemical reactivity) and also have a disproportionate impact on compactibility by increasing cushioning of a material.

According to Figure 12a and Figure 13a, the No.3 Sand from ALS looked to satisfy criteria from the three different bioretention media guidelines, fitting especially well with FAWB recommendation. These PSD guidelines were used as a starting point to estimate the sizing of aggregates required to produce acceptable hydraulic conductivities and hence water drawdown times. ALS No.3 Sand was selected as the base aggregate sand for hydraulic testing.

Figure 15 and Figure 16 show the variation in PSDs of the same materials sourced over three months apart. Results show there are significant changes in PSDs for aggregates and CM2 between March and June. On the other hand, there are only small variations for the composts and CM1 over three months. The result could be indicative of the nature of quarried aggregates; sand from one quarry is bound to be different from sand from another quarry, and may even vary within a quarried deposit. It is simply chance as to which shipment will make it to a particular supply yard on any particular day.

Figure 12a PSDs of aggregates sampled (cumulative limits)

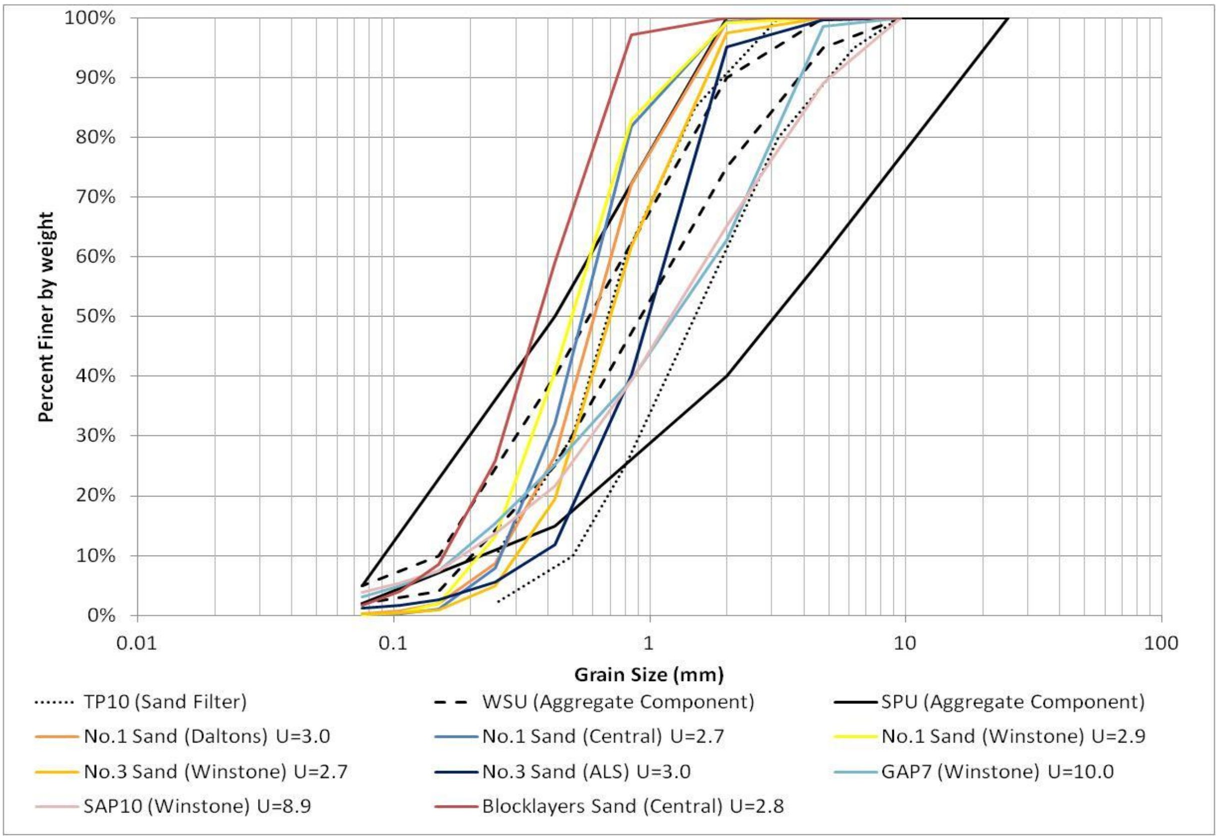


Figure 12b PSDs of aggregates sampled (cumulative limits)

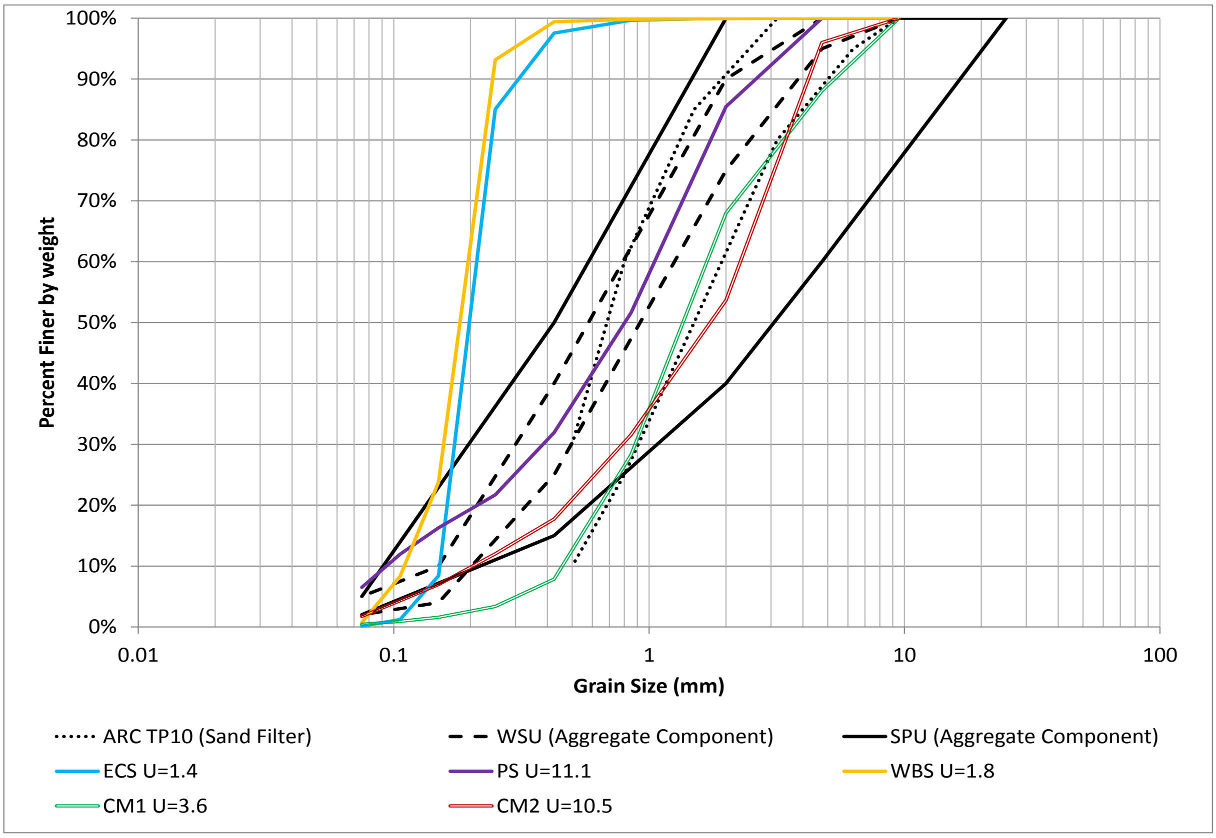


Figure 13a PSDs of aggregates sampled (non-cumulative bands)

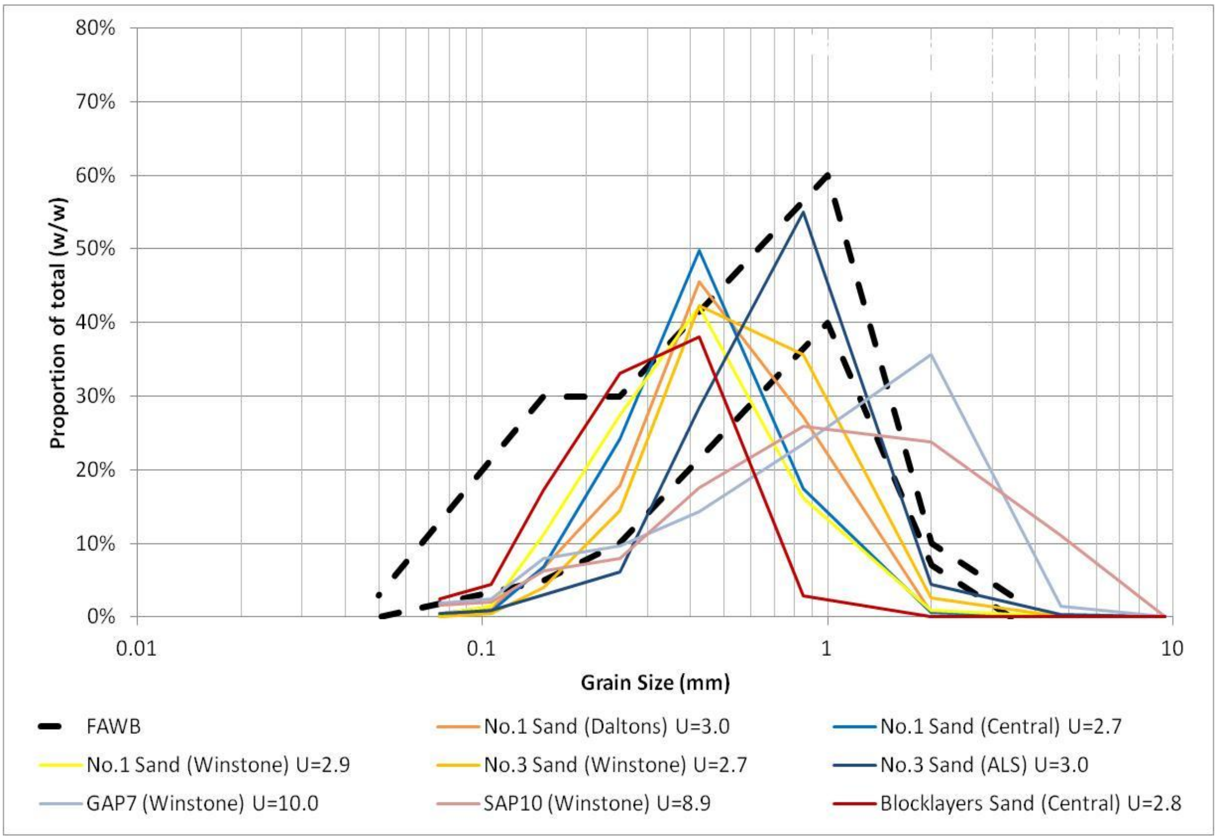


Figure 13b PSDs of aggregates sampled (non-cumulative bands)

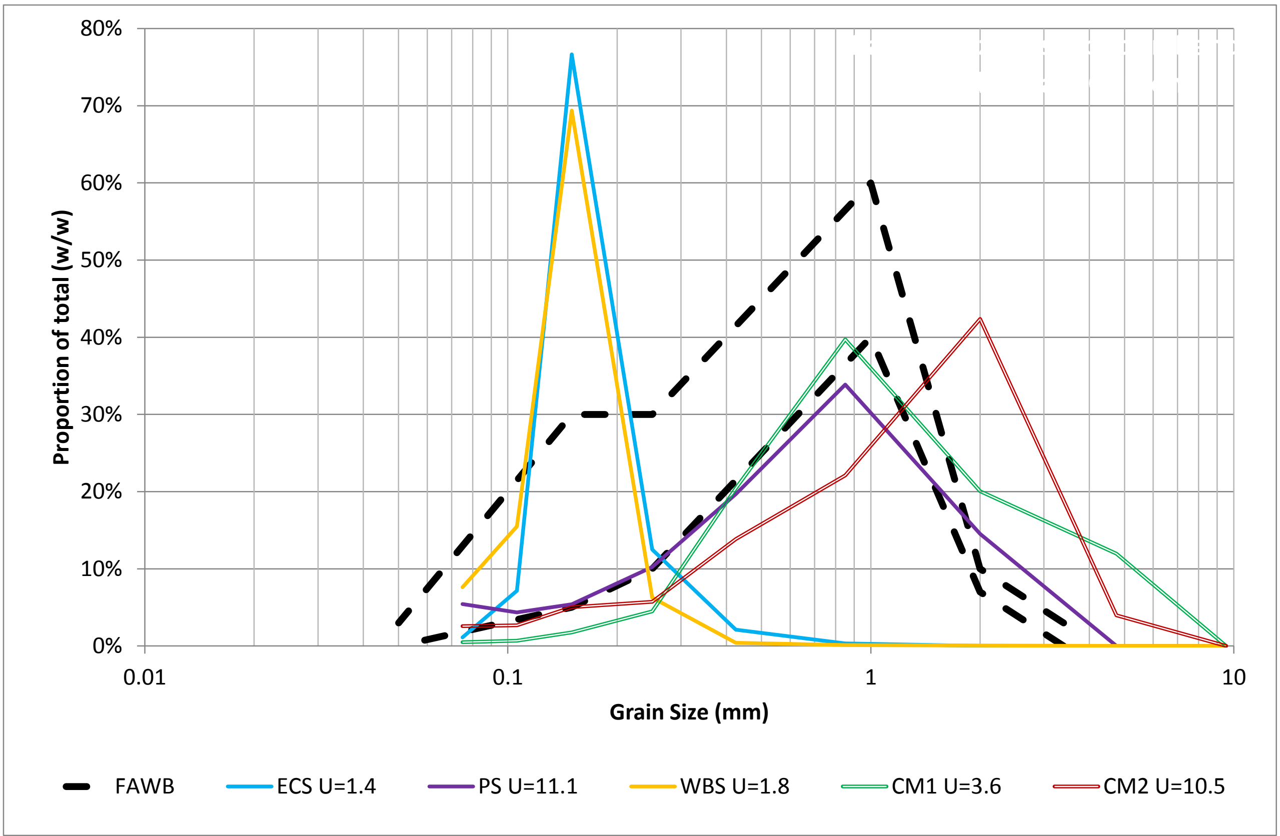


Figure 14 PSDs of composts sampled

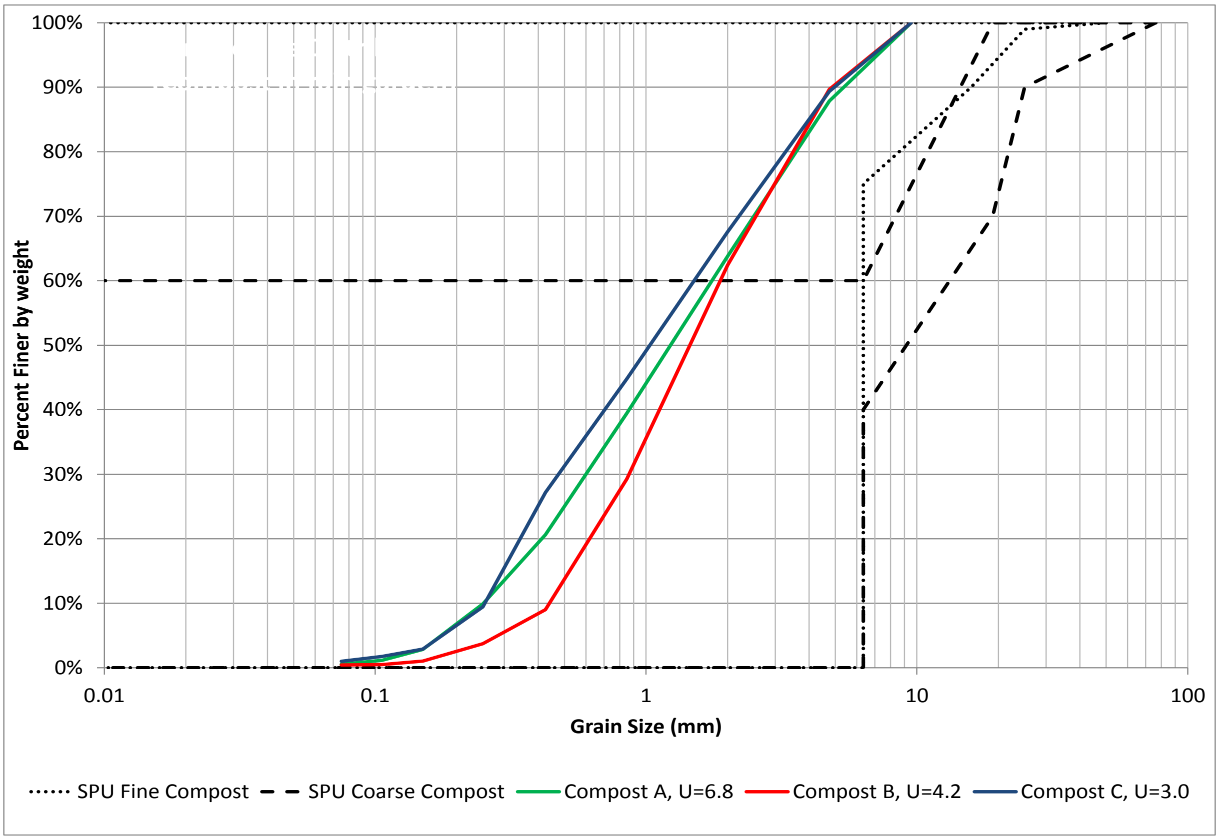


Figure 15 Variation in PSD of aggregates sampled three months apart

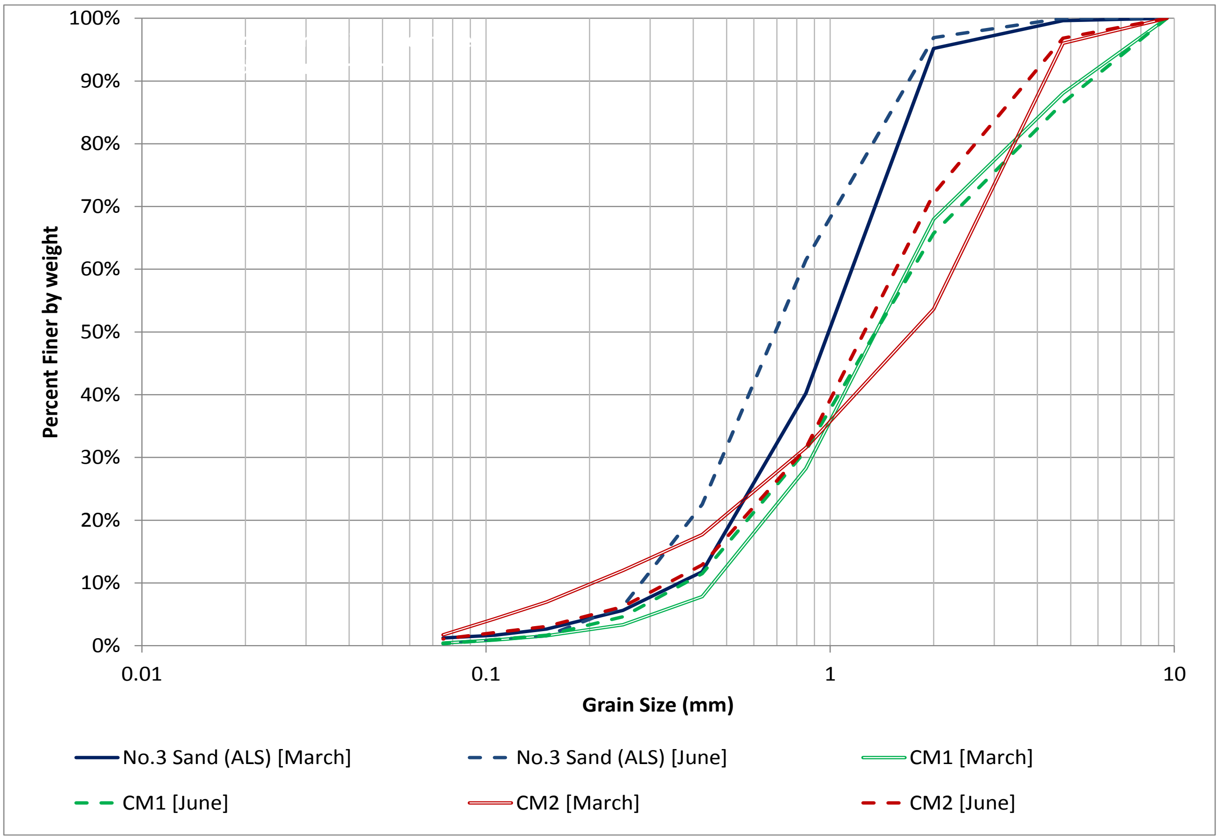
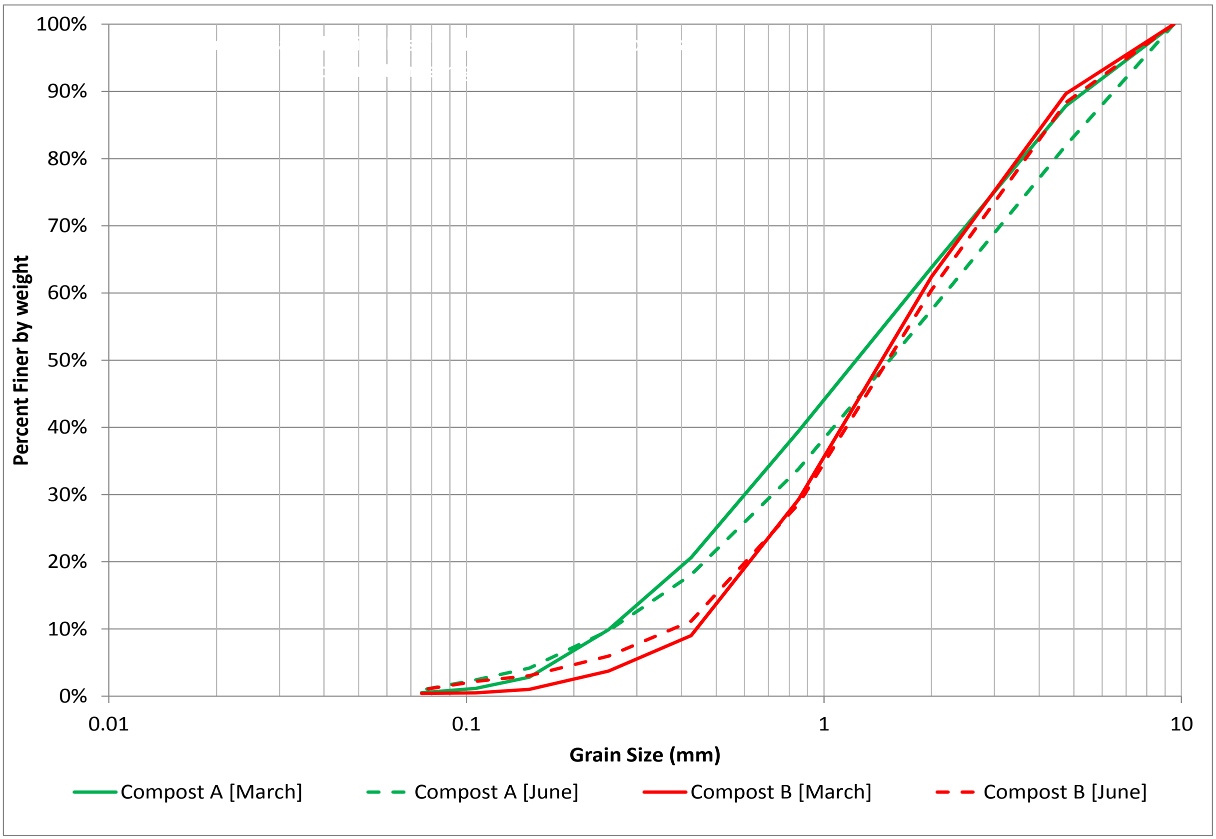


Figure 16 Variation in PSD of composts sampled three months apart



5.3 Compaction testing

Figure 17 shows the behaviour of different media under a constant level of compaction at varying water contents. Moisture density compaction curves test for the maximum dry density to which a particular media can be compacted. A higher density for a material of particular mass is reflective of less pore space, and hence a lower hydraulic conductivity. Smaller pores should also increase storage of plant-available water, which also enhances runoff retention.

Media in bioretention cell applications are not necessarily compacted to the maximum possible level. Nonetheless, a useful measure for designers specifying the level of compaction required is to specify compacting to a certain percentage of maximum dry density. As the density is controlled, the hydraulic conductivity of the media is also controlled. A drawback of using the percentage of maximum dry density specification is that it is specific for each media, and likely must be altered for every different media used.

The two commercial media containing high organic contents displayed a clear parabolic curvature with varying water content (Figure 17a). This suggests variations in water content will have a significant effect on the final level of compaction under a constant level of effort.

The highest density can be achieved for a constant compactive force at the 'optimum' water content, which is 24% for CM2 and 27% for CM1. At a water content of 24%, CM2 is 14% more densely compacted than the CM1. The main reason for the result is the finer texture of CM2 (18% passing 0.425 mm) when compared to the CM1 (8% passing 0.425 mm) (Figure 12b). Altering the water content of CM1 from 37% to 24% can cause a 26% increase in maximum compacted density. Such a substantial change in density is likely to cause significant change in hydraulic conductivity. As both CM1 and CM2 require mechanical compaction to reach target bioretention hydraulic conductivities (Section 5.5), the importance of precise compaction is highly relevant.

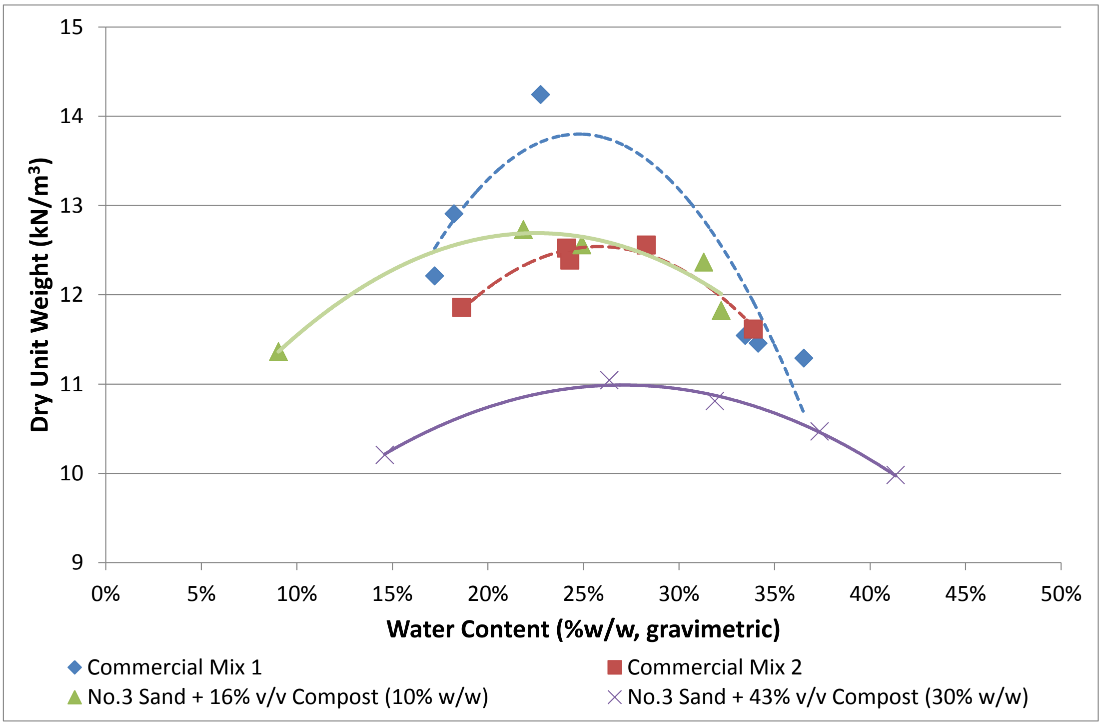
Figure 17b investigated the influence of compost (organic material) proportion on the compaction curve. The parabolic curve for the 16% and 43% (v/v) compost media were not as sharp as the compaction curves for CM2 and CM1, indicating the density of predominantly sand based media are less susceptible to the effects of compaction and water content. However, the shapes of the compaction curves are weakly parabolic, and did not change between the two proportions of compost. Increasing water content of the 16% compost mix by 13% caused a 12% increase in compacted density. For the 43% compost mix, a change of 15% water content resulted in a 10% difference in compacted density.

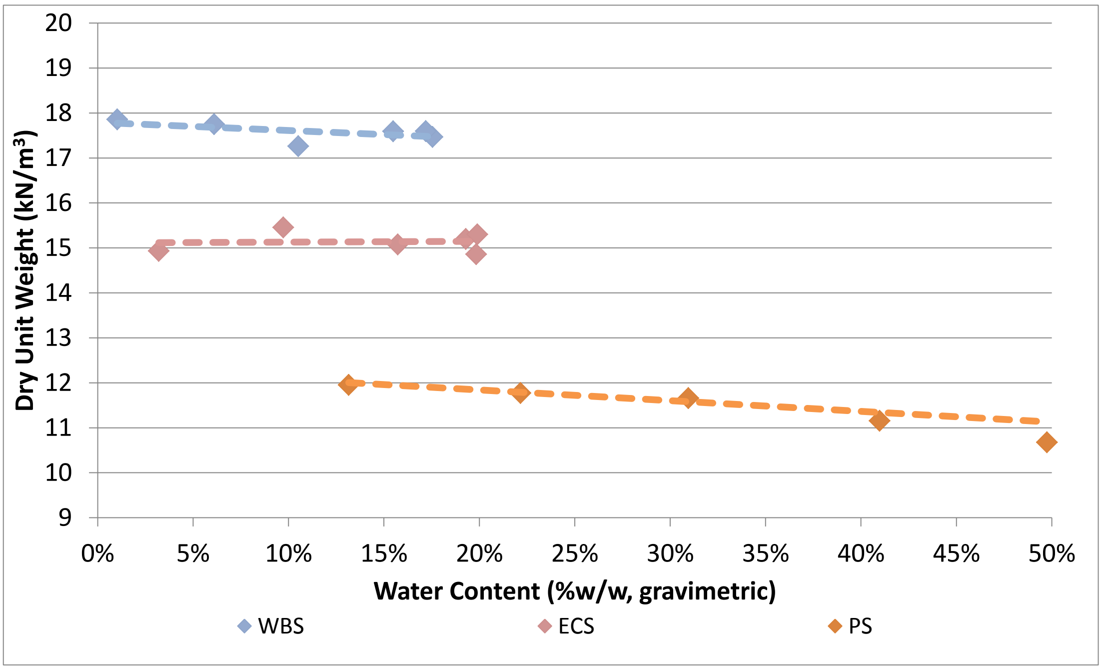
Testing of sands without any compost found ECS and WBS under compaction do not vary with moisture content (Figure 17c). Knowing that sands with 16% (v/v) compost are susceptible to changes under compaction, while sands without compost are not, it can be concluded the organic material is the defining property in sand:organic media which causes the parabolic shape in the compaction curve, indicating variation in density with water content. However, if fine materials (clay/silt) are an appreciable part of a medium, they are also likely to be the defining property (e.g. results of the two CM media).

The pumice sand clearly shows a negative linear trend with increasing water content. The trend is unique among the media tested, and is most likely due to pumice (a brittle sand) being crushed during testing. As water content increases during any compaction test, air voids are replaced with water, and the water particles are able to absorb some of the compactive effort which would have otherwise been distributed on the soil particles. This results in less impact force upon the soil particles. As pumice is crushable, the lowering of impact force on pumice particles results in less crushing, and hence less fine material. A lower amount of fine material ultimately hinders the ability for the pumice particles to re-orient into a denser configuration under compaction. Water content is therefore somewhat important for compaction of pumice, even without an organic material fraction.

Testing has shown compaction is an important consideration for bioretention filter media if the filter media contains 16 - 43% v/v organic material. If specific compaction details (such as water content) are not designed for, filter media may easily be over or under compacted, leading to an undesirable hydraulic conductivity and potential failure of the bioretention cell. An advantage of materials that are relatively insensitive to compaction is a greater certainty of achieving design conductivity range.

Figure 17 Moisture density curves: a) rain garden mixes; b) sands only





Media with a higher proportion of compost are expected to have a higher water holding capacity due to high organic contents. Water holding capacities can be greater than 100% when the water held in the pore spaces weigh more than the soil particles themselves. The potential for high water content upon delivery enhances the importance of accurate compaction instruction, as a small change in water content can cause a significant difference in compacted density (Figure 17a) and therefore final hydraulic conductivity. Compaction issues relating to variable or unpredictable media water content can be prevented by testing the water content before installing the media, and adjusting the compaction process correspondingly. Perhaps a more practical option is to eliminate the variability by controlling the water content. An example of a simple control is keeping the media dry by storing it under shelter, drying wet media before mixing and/or not mixing media during rainy weather[3]. These controls are particularly important for the compost component. It is also recommended to ensure bioretention systems are 'off-line' during construction. Beyond compaction, water content showed significant effects on the mixing process for engineered media in living roof applications, resulting in quite substantial deviation in overall composition from the intended specification (Fassman et al. 2010). Similar mixing issues could occur for bioretention applications.

5.4 Laboratory replication of field compaction

The 2009 bulk density of Corban Ave (adjacent to Oteha Valley Road Park-n-Ride in Albany) bioretention cells constructed in 2006 using CM1 was 1.07 g/cm^3 (Landcare Research, unpublished data). In order to achieve this density with 600 mm of media depth (9,236 cm^3 with a 140 mm diameter column), 9,883 g of CM1 is required to be compacted into the volume. Following TP10's guideline of construction (ARC 2003), it took 15 blows of a modified proctor hammer (2,700 kN-m m^{-3} force or 4.5 kg falling through 457 mm) on two 300 mm lifts to achieve the required density. A replicate test was also performed confirming the result. Therefore the equivalent of light tamping with a backhoe bucket is 15 blows with a modified proctor hammer.

5.5 Hydraulic conductivity testing

Aiming for an optimal 2 to 24 hr drawdown time from 300 mm ponding depth equates to hydraulic conductivity of 12.5 to 150 mm hr^{-1}. Hydraulic conductivity within this optimal

[3] Excavation during rainy weather can hinder infiltration performance, as "smears" the bottom and sides of the excavated area, reducing the potential for infiltration (Brown and Hunt 2010).

range would be the most likely to provide adequate media to runoff contact time for pollutant removal.

The falling head permeability method (section 4.5) was preferred in order to simulate field conditions as closely as possible. Figures 18-21 each compiles results from at least 10 falling head permeability measurements on specific columns. The results show a high level of random variation between test runs, especially on highly permeable media. Maximum difference between tests on a single column was up to 2000 mm hr^{-1} and up to 150% relative difference between minimum and maximum permeabilities, however this reduced significantly for less permeable media.

Figure 18 shows the effect of compaction on two commercial mixes (with high proportion of organic material) as well as Pumice sand + 10% v/v compost mix and a No.3 sand + 10/12% v/v compost mix. For the two commercial mixes, hydraulic conductivity without mechanical compaction (wetting only) was five to seven times greater than the upper limit of the optimal range (830 mm hr^{-1} and 1,800 mm hr^{-1} for CM1 and CM2 respectively, compared to the 150 mm hr^{-1} upper limit). It is assumed that these two proprietary mixes were designed with the TP10's compaction requirement of light tamping with backhoe bucket (ARC 2003) in mind. Indeed compaction by light tamping caused hydraulic conductivities to decrease by 75%. With compaction, these two mixes reach hydraulic conductivities close to the 150 mm hr^{-1} upper limit of the optimal range without factoring clogging (180 mm hr^{-1} and 240 mm hr^{-1} for CM1 and CM2 respectively), while variations between test runs decreased and hence hydraulic performance was more consistent.

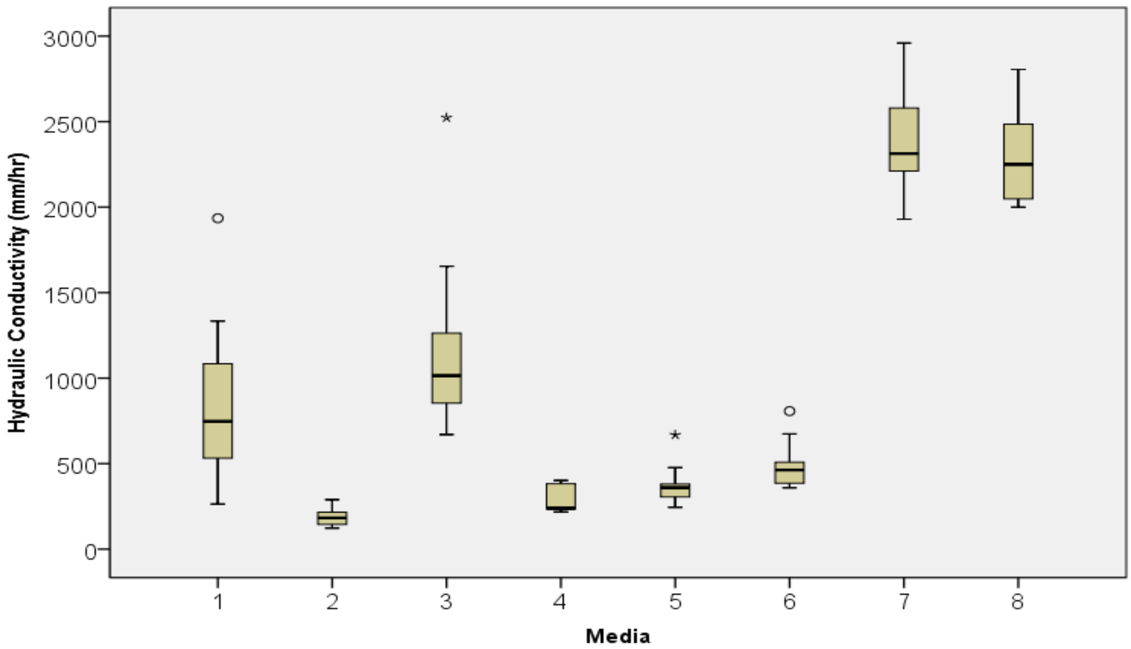
The Pumice Sand + 10% v/v compost mix showed no significant difference in hydraulic conductivity between columns compacted by wetting and columns compacted by light tamping (based on an independent sample Mann-Whitney U Test, p-value = 0.179). The potential crushing of pumice sand observed when determining the moisture-density curves was not realized through the light tamping or wetting compaction process.

Similarly, No.3 Sand + 10 or 16% v/v compost mix also showed no significant difference between the two compaction conditions (p-value = 0.596). The difference in the volume of compost arise from changing testing from proportion by weight to proportion by volume (see section 4.5). The results indicate the hydraulic performance of sand based mixes is resilient to variation under light tamping compaction, which was not the case for the high organic content commercial mixes.

Figure 19 shows the best fitting aggregate (No.3 Sand) according to PSD guidelines (Figure 12a) resulted in an extremely high hydraulic conductivity (2,310 mm hr^{-1}) As demonstrated in Figure 18, mechanical compaction on No.3 sand based mixes does not lower the hydraulic conductivity due to the sand component being resilient to compaction.

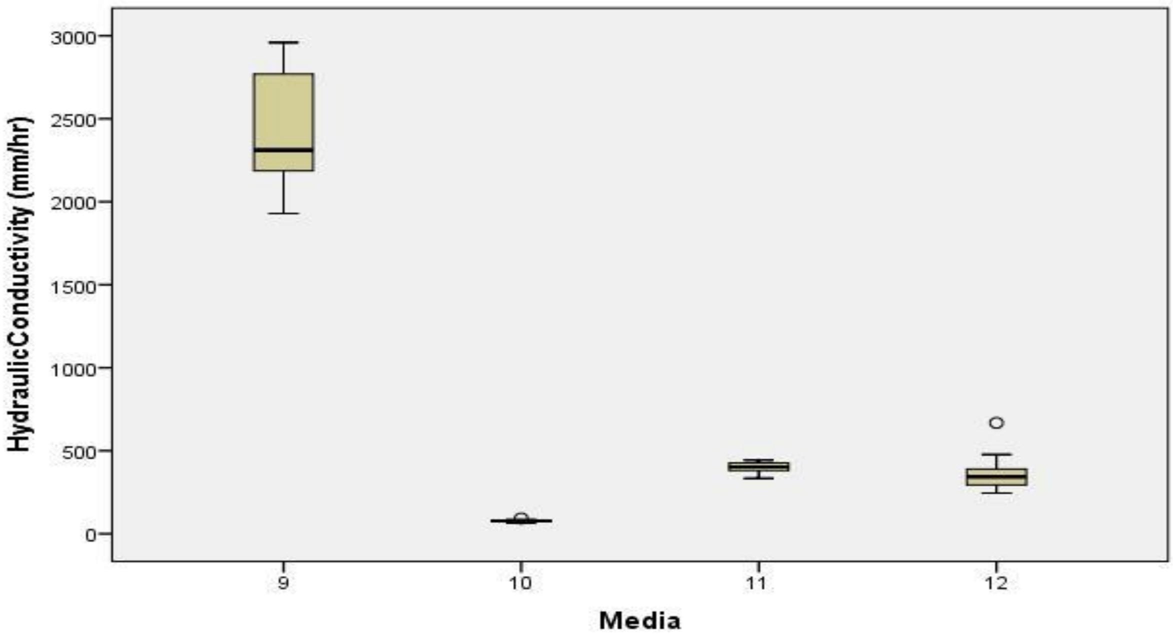
The 90% East Coast Sand + 10% Compost B was the most successful sand in terms of satisfying hydraulic conductivity objectives (80 mm hr^{-1}). The result was surprising because East Coast Sand is fine and poorly graded; arguably the poorest fitting of all sands according to the international criteria (Figure 12b). Woodhill Black Sand + Compost B and Pumice Sand + Compost B (each 90% : 10% v/v; 400 mm hr^{-1} and 340 mm hr^{-1}, respectively) were the next closest mixes to the optimal range. Woodhill Black Sand is similar to East Coast Sand in terms of PSD and overall grading, and therefore also poorly fitting the PSD guidelines. Pumice Sand is a fairly good fit for the coarse section of the PSD guidelines (greater than 0.2 mm), has a greater amount of fines than recommended, but is well graded.

Figure 18 Effect of compaction on saturated hydraulic conductivity of bioretention filter media mixes



1 = CM1 [wetting]
2 = CM1 [lightly tamped]
3 = CM2 [wetting]
4 = CM2 [lightly tamped]
5 = 90% PS + 10% (v/v) compost B [wetting]
6 = 90% PS + 10% (v/v) compost B [lightly tamped]
7 = 84% No.3 Sand + 16% (v/v) compost B [wetting]
8 = 90% No.3 Sand + 10% (v/v) compost B [lightly tamped]

Figure 19 Effect of aggregate type on saturated hydraulic conductivity of bioretention filter media mixes (all compacted by wetting)



9 = 90% No.3 Sand + 16% (v/v) compost B
10 = 90% ECS + 10% (v/v) compost B
11 = 90% WBS + 10% (v/v) compost B
12 = 90% PS + 10% (v/v) compost B

All three of the relatively “successful” sands have higher percentage fines than the recommended guidelines, while two out of three are poorly graded. All three have extremely different PSDs compared to the “best fitting” No.3 Sand. With this result, it is clear the PSD-based guidelines should not be used as a substitute to hydraulic conductivity testing.

While No.3 Sand was tested with a slightly higher proportion volume of compost (12% compared to 10% for the remaining mixes), this slight difference is not expected to have a substantial impact on the result since the sand-based mixes are so resistant to compaction. The difference between the results is attributed to the highly different PSDs of the aggregates.

Figure 20 shows the effects on hydraulic conductivity of varying media and ponding depths, different proprietary mixes, different composts and different proportion of compost. TP10 specifies a 1,000 mm media depth and 220 mm ponding depth, as opposed to the 600 mm media depth and 300 mm ponding depth tested on most columns. The CM1 was randomly selected to test hydraulic conductivity under both specifications. No significant difference was found between the 1000 mm media depth column and the 600 mm media depth column (independent sample Mann-Whitney U test, p-value = 0.199); however neither depth limited hydraulic conductivity to the optimal range. Neither CM1 nor CM2 with 600 mm media depth (830 mm hr^{-1} and 1,800 mm hr^{-1}, respectively) met the optimal hydraulic conductivity range.

Further information in Figure 20 shows increasing the proportion of compost in the mix decreases hydraulic conductivity. However at 36% v/v the hydraulic conductivity of the mix is still an order of magnitude higher than the identified criteria. Achieving the sought after hydraulic conductivity by increasing organic matter is undesirable, because organic matter degrades and likely leaches nutrients over time.

It is also clear mixes with Compost B tend to be closer to the required hydraulic conductivity than mixes with Compost A (3,490 mm hr^{-1} for 12% v/v Compost A compared to 2,310 mm hr^{-1} for 16% v/v Compost B). The result was unexpected due to Compost A consisting of more fine material than Compost B. The 4% difference in compost proportion between the two mixes should not be significant enough to account for the major difference in hydraulic conductivity (because of the overall predominance of sand in the mix). It is speculated the difference could be due to the different physical properties of the constituents: bark for Compost B and green waste for Compost A, however further study would be required to investigate the hypothesis, and is outside of the scope of this study.

Few of the mixes met the target range for hydraulic conductivity; however, the range is derived from a best estimate rather than an extensively documented result in the current literature. Ultimately the objective of the research was to examine performance of combinations of readily available materials. Within these constraints, the five mixes (Figure 21) that most closely achieved the target hydraulic conductivities without mechanical compaction were selected for water quality testing. The five mixes and their respective compactions are:

- CM1 [lightly tamped]
- CM2 [lightly tamped]
- East Coast Sand + 10% Compost B [wetting]
- Woodhill Black Sand + 10% Compost B [wetting]
- Pumice Sand + 10% Compost B [wetting]

Figure 20 Effect of other variables on saturated hydraulic conductivity of bioretention filter media mixes

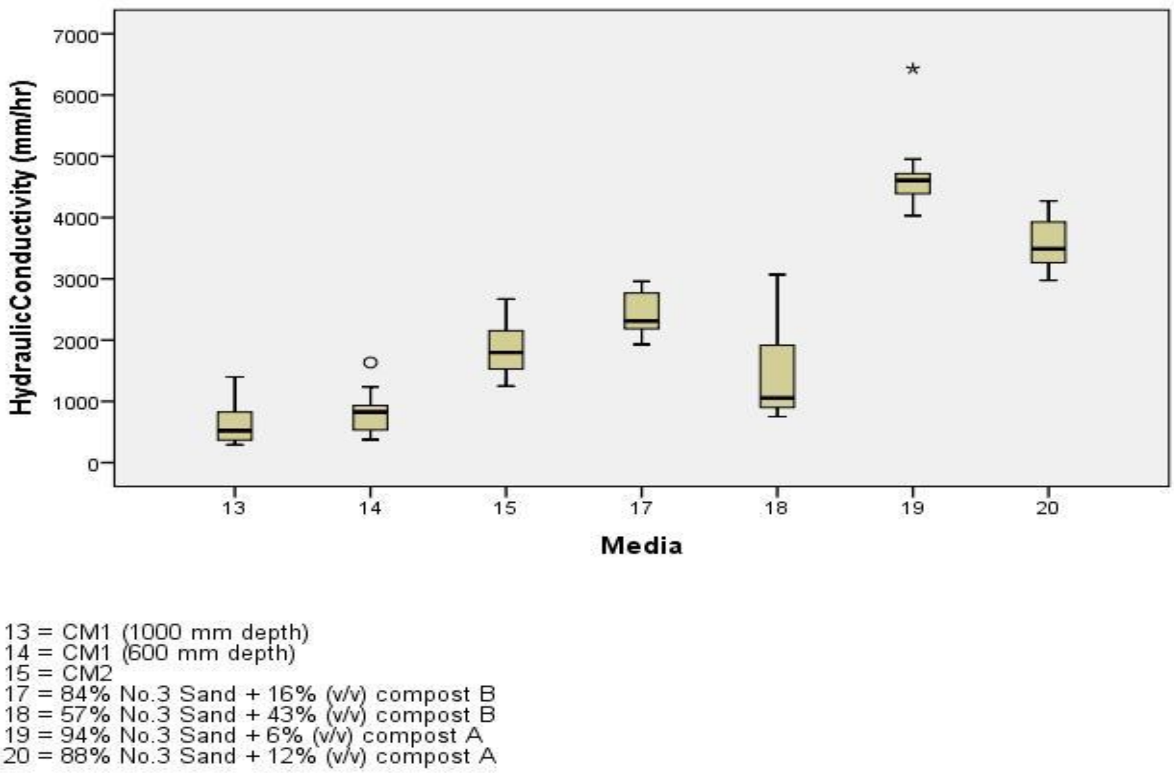
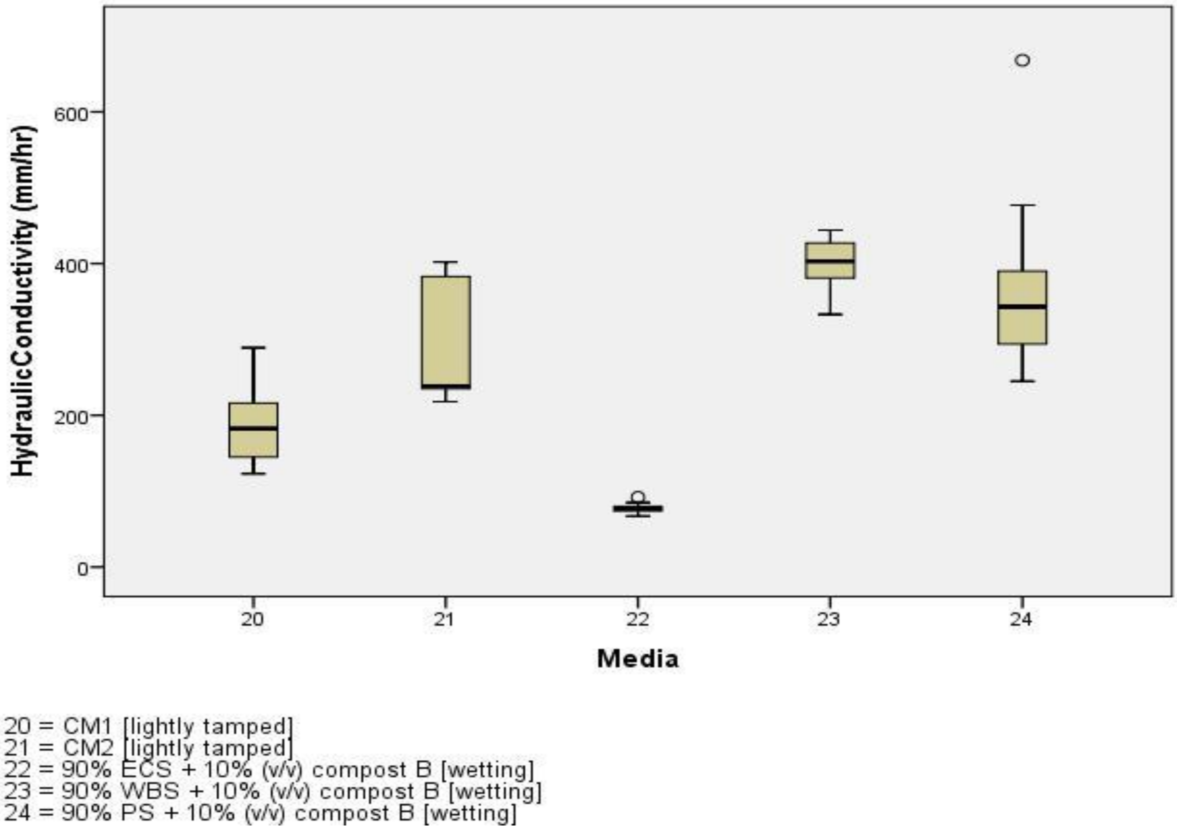


Figure 21 Saturated hydraulic conductivities and compaction method of bioretention filter media mixes to progress to water quality testing stage



5.6 Chemical analysis of material

5.6.1 Individual Components

Table 18 provides information on the actual content of organic matter within each media trialled, as it is this characteristic of a media that contributes most to plant viability, influences discharge quality, and likely ranges substantially between different sources of compost. Washington State and SPU have recently reduced the maximum allowable organic content of compost from 10% to 4 - 8% w/w (determined by loss on ignition test) as the former was "too high" for nutrient management (Hinman, personal communication 2011). Both of the commercial mixes exceed this guideline, while the low organic content of the non-proprietary mixes leads to additional investigation of actual nutrient storage and supply against plant needs.

Table 19 shows the results of chemical analysis of individual media components investigated, divided into organic and sand components. The Compost A and B have similar pH (above 6.0) and about 25% v/v total carbon content. The bark base of Compost B may be reflected in a slightly higher C:N ratio (30) than the green waste-derived Compost A (C:N 25); bioretention mixes with both composts are likely to require small amounts of nitrogen to allow plants to grow quickly during establishment. Both composts had very high total phosphorus and Olsen Phosphorus concentrations; this combined with C:P ratios below 100 and low anion retention (a measure of AEC) indicates both composts are likely to release phosphorus. However, this can be mitigated by combining the composts with sands or other amendments with phosphorus-mitigating chemistry. For example, as both composts have high calcium and magnesium concentrations, there is potential to form insoluble phosphate precipitates at a neutral or higher pH[4], particularly with iron- or aluminium- compounds (such as water treatment residuals [O'Neill and Davies 2012 a, b]). There might remain a risk of limited ability to reduce phosphorus of incoming stormwater runoff if the media is acting to control itself. In the absence of clays, the compost component was expected to be the major contributor to nutrient storage and buffering, as measured by CEC. All organic components had CEC in excess of 60 cmol(+)/kg, indicating they would contribute this important function for plants.

[4] Very limited data for Auckland suggests that road runoff pH ranged 6.3-7.5 on one an arterial road through a residential area (Fassman and Blackbourn 2011), and showed a mean pH of 7.2 over 12 events from another road through a light industrial catchment (Auckland Regional Council, unpublished data).

The strongly acid pH (4.7) and very low nutrient (P, K, Mg) content of Compost C reflects the high proportion of peat in this material. Compost C did have high phosphorus removal potential (high anion retention) and low potential for mineralisation of organic P, however, because this material would require addition of several amendments to increase the pH to over 6 (for metal removal) and allow adequate plant growth Compost C was not used for further development in this study but with relatively minor amendments is likely to be suitable for bioretention media.

All sands were expected to contain negligible organic matter and plant-available phosphorus. The compost component is therefore crucial towards providing the carbon, nitrogen and phosphorus required to adequately sustain plant life during the bioretention establishment phase. In the medium and long term most bioretention systems are expected to receive adequate nutrients in stormwater runoff.

The sands (aggregates) tested had a variety of mineralogies, ranging from iron sand to volcanic materials (pumice and scoria) to beach sand (East Coast Sand which has a shell component). Mineralogy can influence the ability of the sands to contribute to pollutant removal and buffering. All aggregates had a neutral to basic pH. Scoria and Woodhill Black Sands had pH around 8. For aggregates, the East Coast Sand and SAP10 have surprisingly beneficial levels of anion retention, which indicates phosphate retention potential. The East Coast Sand is likely to have the greatest capacity to immobilise metals and phosphorus due to a combination of high pH and exchangeable Ca, reflected in a moderate anion retention (50%). Scoria was the only other aggregate with the potential to attenuate phosphorus through forming iron phosphates, which are extremely insoluble. This is reflected in the low Olsen Phosphorus of 5 despite a very high Total Phosphate concentration. Scoria sand contributed to CEC with 13 cmol(+)/kg. Unfortunately, although scoria was identified as a desirable aggregate, meeting with the supplier revealed that scoria production in the Auckland region in the future is uncertain, with no substantial new resources being available once Three Kings Quarry is exhausted (Winstone's pers. comm. 2010). Scoria should be considered for inclusion in bioretention substrate in areas where local supplies are available. All the remaining sands (East Coast, Pumice and Woodhill) were selected to mix with compost in plant growth and leaching trials.

Table 18 Organic content in media mixes trialled for water quality

Media	Organic Content (% w/w)
90% Woodhill Black Sand + 10% Compost B	0.46
90% Pumice Sand + 10% Compost B	1.81
90% East Coast Sand + 10% Compost B	0.88
CM1	14.36
CM2	16.94

Table 19 Chemical analysis of individual materials

Material	pH	Organic C (%)	Total N (%)	C:N ratio	Olsen P (mg/kg)	Total P (mg/kg)	Anion retention (%)	C:P ratio	CEC (cmol(+)/kg)	Base CEC saturation[2] (%)
Compost C	4.7	17.7	0.50	35	8	269	97	657	61	10
Compost B	6.2	28.8	0.97	30	393	4,080	9	70	70	174
Compost A	6.6	23.5	1.10	21	259	2,660	21	88	62	137
Pumice Sand[1]	7.2	N/A	N/A	N/A	N/A	75	1	N/A	N/A	N/A
Woodhill Black Sand[1]	7.9	N/A	N/A	N/A	N/A	445	2	N/A	N/A	N/A
East Coast Sand[1]	8.9	N/A	N/A	N/A	N/A	135	53	N/A	N/A	N/A
No.1 Sand (Winstone)	7.0	0.03	<0.01	N/A	2	266	4	1	3	72
SAP10 (Winstone)	8.1	0.10	<0.01	N/A	5	2,220	24	0	13	94

1. N/A indicates not available because material was tested only in a blend. Many of the parameters were not tested as individual materials as characteristics in blends were of greater interest for field applications (Table 19).

2. Base saturation is a percentage of the CEC value. Substrates with low CEC are easily saturated; low concentrations of cations are needed to exceed 100% base saturation.

5.6.2 Bioretention Mixes

The commercial mixes with a high proportion of organic material (>25% v/v) have significantly higher CEC compared to mixes with only 10% (v/v) compost (Table 20a). Compost is probably the vital component for providing the CEC required for heavy metal removal in these mixes, assisted by a higher clay and silt content providing larger active surface area. The Pumice Sand mix and SAP10 (Table 19), being derived from volcanic materials, do contribute some minimal chemical pollutant removal via slightly elevated CEC, while East Coast Sand and Woodhill Black Sand mixes have close to zero CEC. However the East Coast Sand has a relatively high anion retention, similar to both commercial mixes. The mixes with close to zero CEC and anion retention have no chemical pollutant removal mechanism. The importance of sand in these cases is to physically stabilize the system and provide an adequate water retention time for the compost components to perform chemical pollutant removal, as well as providing physical filtration to remove contaminants attached to the filtered sediment. CEC of the sand mixes could be boosted, if required, by addition of zeolite, or material containing allophanic, aluminium, or iron oxides. The latter approach is used in North Carolina where a relatively uniform product is available (for use in baseball pitches), while aluminium-rich drinking water treatment plant residuals are under investigation by WSU (Hinman, personal communication, 2012) and the University of Maryland (O'Neill and Davis 2012 a, b).

The three non-proprietary bioretention mixes had pH above 7; the addition of 10% (v/v) compost lowered the pH of both East Coast Sand and Woodhill Black Sand by about half a pH unit, but did not change the pH of Pumice Sand mix (Table 19 for individual components vs Table 20a for mixes). The latter result was probably due to slightly higher buffering reflected in CEC of 6 cmol(+)/kg base. All three 90% sand: 10% compost mixes had low to very low CEC. The slightly higher Pumice Sand mix result is consistent with 1% higher Total Carbon than the two other sands, but the result was unexpected as all three sands were assumed to have a negligible organic content before mixing with compost. The Total Carbon test was repeated, with a similar result (Table 20b, leached samples). Carbon can be present within Pumice deposits as charcoal (from trees incinerated during pumice deposition or from organic material in alluvial deposits). A follow-up with the supplying quarry is needed to identify the cause and see if the result is consistent through time.

The two commercial bioretention mixes were similar to the East Coast Sand mix in only one respect: having moderate anion retention. Unlike all sand mixes the commercial mixes had a high CEC, total carbon content, Total Phosphorus and Olsen Phosphorus concentrations. This indicates superior qualities for plant growth, however, the lower C:N ratio of the CM1 indicates a greater potential for mineralisation of organic component, which is seen as discolouration of leachate and elevated Dissolved Organic Carbon (DOC) (Figure 22). DOC is implicated in forming soluble metal complexes. Countering this, the pH of the CM1 was neutral and about 1 pH higher than the CM2. At pH 6 metals are likely to be more soluble, and Calcium-phosphate complexes will not precipitate, i.e. the higher Ca concentrations of CM2 cannot contribute to removal of soluble phosphate. Bioretention substrates may gradually acidify over time, as nitrogen and sulphur oxides in vehicle emissions are acidic and input of organic matter also tends to be slightly acidifying (and vary with plant species), although Auckland's limited road runoff data suggests near neutral pH (see footnote in Section 5.6.1). Mulch and near-surface soil pH can be increased post-installation using readily available amendments if required, but a recommendation is premature at this stage.

Table 20a presents results from unleached samples and a hand-mixed commercial sample (mixed while visiting the supplier); Table 20b presents results from samples leached with water including bulk samples of both commercial mixes, with a focus on availability and sources of phosphorus (four tests of phosphorus are presented in Figure 20b). The commercial mixes originated from minimum 1 m^3 bulk samples; all mixes were placed in columns and leached with four equivalent volumes of tap water over two days before being dried and the media analysed. Organic matter levels in leached samples are generally higher as the short leaching time did not allow its mineralisation. Although dissolved organic matter was washed out (creating discolouration of samples), this does not impact the volume of organic matter. The organic component is coarser and less vulnerable to leaching compared to the silt and clay components; as the fines component of the CM1 and CM2 was much greater than in the three sand mixes, CM1 and CM2 lost a disproportionate amount of fines.

The leaching procedure would expect to cause excess nutrients to wash out, if present. However, if nutrients are in substantial excess, they would still be detected in a subsequent analysis of media chemistry. The analyses indicate fertiliser phosphorus is likely to have been added to both of the commercial rain garden mixes, whereas it was not added to the hand-mixed sample of CM2 analysed in Table 20a. About 20 to 25% of the phosphorus in all mixes is in organic form and likely readily mineralised due to low C:P ratios (ratios below 200 generally indicate mineralisation of organic phosphorus will occur, especially when C:N ratio are low). Eliminating suspected added fertilizers in commercial mixes would mitigate some concerns for water quality performance (confirmed by results in Section 6), while the CM2 would also benefit from increased pH.

Figure 22 Effluent turbidity in laboratory water quality testing trials.

ECS+	WBS+	PS+	CM2	CM1
1.5 NTU	22.7 NTU	7.6 NTU	2.1 NTU	73.2 NTU

Table 20a Chemical analysis of Stage 2 mixes: unleached

Material	pH	Organic C (%)	Total N (%)	C:N ratio	Olsen P (mg/kg)	Total P (mg/kg)	Anion retention (%)	Exch. Calcium (cmol(+)/kg)	CEC (cmol(+)/kg)	Base CEC saturation[2] (%)
CM1	7.1	6.3	0.44	14	69	1,100	52	28.7	32	140
CM2[1]	6.0	9.5	0.17	57	28	650	51	17.6	25	95
Pumice Sand + 10% Compost B (v/v)	7.2	1.52	0.06	27	26	251	1	6.6	6.3	144
Woodhill Sand + 10% Compost B (v/v)	7.4	0.23	0.03	9	7	472	1	1.8	1.8	156
East Coast Sand + 10% Compost B (v/v)	8.3	0.47	0.03	17	14	189	51	29.8	1.1	3005

1. Sample was a small, hand mixed sample without any fertilisers added.

2. Base saturation is a percentage of the CEC value. Substrates with low CEC are easily saturated; low concentrations of cations are needed to exceed 100% base saturation.

Table 20b Chemical analysis of Stage 2 mixes: leached by water

Material – leached by water	Organic C (%)	Total N (%)	C:N ratio	Total P (mg/kg)	0.5M $H_2SO_4^-$ Soluble P (mg/kg)	Organic P	C:P ratio
CM1	8.35	0.64	12.9	2230	1770	460	37
CM2	9.85	0.29	34.1	1630	1210	420	60
Pumice Sand + 10% CAN fines bark compost	1.05	0.02	52.3	190	190	<10	55
Woodhill Sand + 10% CAN fines bark compost	0.27	<0.01	27.0	390	360	30	7
East Coast Sand + 10% compost	0.51	0.01	46.9	180	140	40	28

6.0 Results: Water Quality Testing

The intention of the testing regime was to enable investigation of typical storm-event bioretention performance, as well as comparative effects as the media is aged (in terms of pollutant loadings). This was achieved by alternating the pollutant dose between WQV storms and concentrated doses simulating multiple years of loadings (Table 14). The schedule allowed for investigation into bioretention performance at 0 year, 5 year, 10 year and 15 year intervals. The dosing pattern and total number of tests was considered a balance between satisfying project objectives in a relatively short time frame, and the practicalities of performing laboratory work.

6.1 Runoff to media contact time

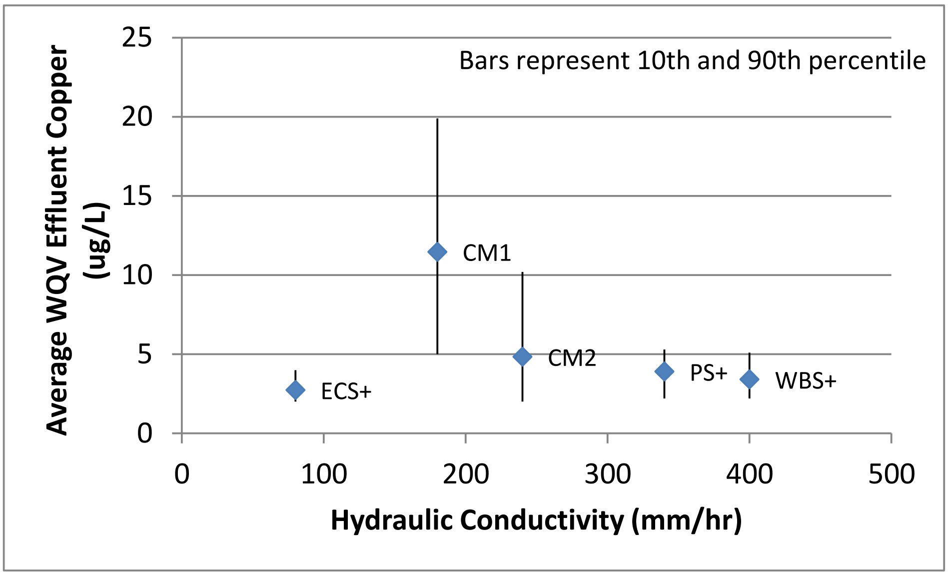
Investigation into a possible link between the runoff to media contact time and pollutant removal efficiency was conducted. The five media subjected to water quality testing were chosen based on target hydraulic conductivities to balance a mix of design criteria. Figure 23 plots the mean of 15 WQV effluent concentrations (WQV doses are typical stormwater concentrations) against the hydraulic conductivities of respective media. Hydraulic conductivity is used as a representation of runoff to media contact time. A low hydraulic conductivity represents a high runoff to media contact time, and vice versa. The scattering of points and lack of a clear trend suggest there is no link between hydraulic conductivity and removal efficiency in any of the three pollutants investigated. Over the range of conditions tested, there is also no direct discernible relationship between hydraulic conductivity and the range of effluent concentrations.

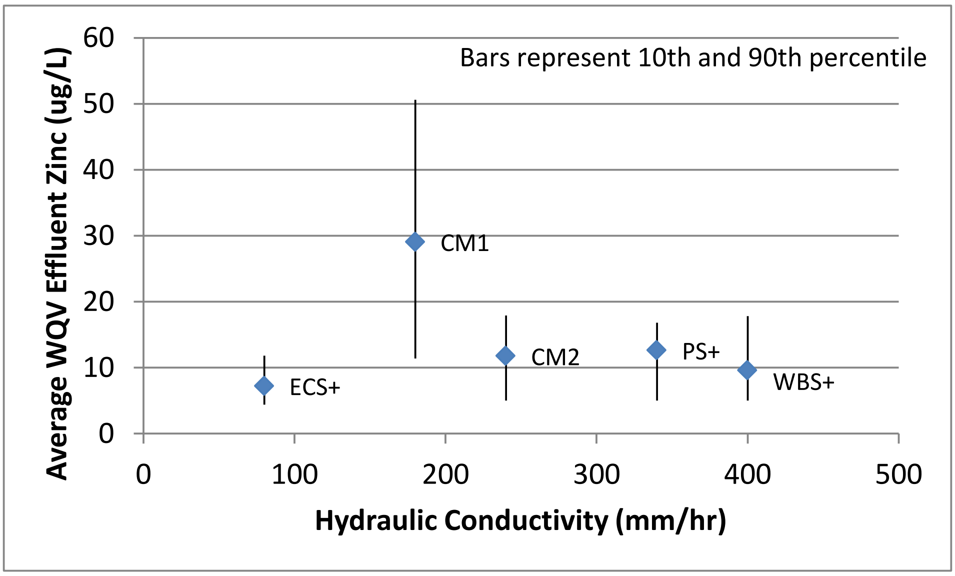
Adsorption is the only pollutant removal mechanism tested in the experiment. Adsorption of cations or anions through exchange sites is known to be a rapid reaction, while specific adsorption is “less rapid” (Lucas & Greenway 2011). The hydraulic conductivities of media tested are diverse, ranging from 80 to 400 mm hr^{-1}. If the time required for specific adsorption to take place is within the range, the results should show media with lower conductivity performing better than media with higher conductivity due to the adsorption having enough time to take place. Figure 23 shows no correlation between hydraulic conductivity and quality of effluent. For example, CM1 has the second lowest hydraulic conductivity, yet produces effluent with the highest heavy metal concentrations, and the lowest phosphorus.

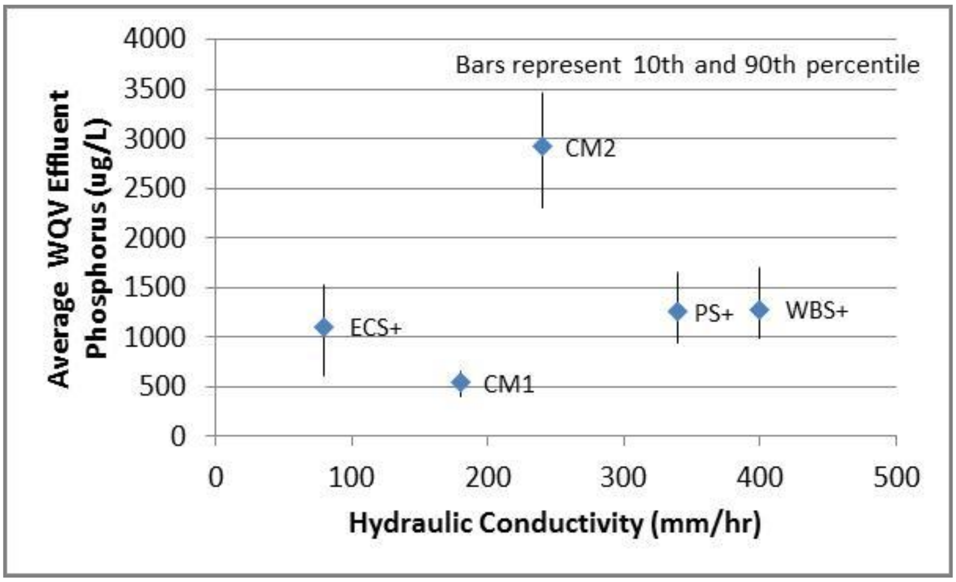
The results suggest runoff to media contact time is not a factor in pollutant removal efficiency over the tested range of hydraulic conductivities, media, and pollutants. This confirms that the less rapid form of specific adsorption should have taken place within the entire tested hydraulic conductivity range. The inherent chemical properties of the media, such as CEC, base CEC saturation, anion exchange capacity (AEC)(aka anion retention), and mineralogy are likely to be more determinant of the pollutant removal efficiency.

Despite the result indicating hydraulic conductivity does not affect pollutant removal efficiency in laboratory conditions, it is likely to play a larger role in real bioretention applications. A low hydraulic conductivity ensures runoff is retained in the filter media or temporarily retained as ponding. It provides greater opportunity for infiltration into sub-soils, hence reducing volume of runoff discharged as well as the pollutant load entering aquatic environments. Runoff volume reduction has been found to be a large factor in removing pollutant mass in field conditions (Brown & Hunt 2011b; DeBusk & Wynn 2011; Carpenter & Hallam 2010). Lower hydraulic conductivity filter media are therefore preferable to higher hydraulic conductivity, as long as conditions for draw-down are met. Pollutant removal by runoff volume reduction was not reproduced in this study.

Figure 23a-c Relationship between hydraulic conductivity and effluent pollutant concentration







6.2 Cumulative pollutant mass loading and breakthrough

Figure 24a and Figure 24b show the cumulative influent versus cumulative effluent loads over the entire testing period for copper and zinc respectively. Cumulative loading results should be recognised as only a rough estimate of how different media may perform in pollutant removal over the long term. The data assume 100% of influent volume becomes effluent; i.e., no runoff is held in the media. Plants are also not simulated for this laboratory study, nor is microbiological activity associated with plant roots (i.e. in the rhizosphere). The ability of plants to uptake heavy metals has been found to be up to 10% of total heavy metals removed (Davis et al. 2001).

Consistent, non-deteriorating performance during the simulated 5 year and 1 year aging processes is also assumed; in other words, the pollutant removal performance was assumed to be the same from the beginning of the 5 year aging period, to the end of the 5 year aging period (and same applies for the 1 year aging periods). Increased influent concentrations may inflate sorption potential for the concentrated doses.

Over 15 years, copper mass loads (Figure 22a) are estimated to be reduced by around 60% for the sand based mixes, and 36% for CM2. CM1 is estimated to export 15% more mass than was inputover a 15 year period. Overall leaching of copper indicates there was a substantial amount of copper already present in the CM1 at initial installation. Possible reasons for copper leaching in CM1 are discussed later in this section. Further examination of how CM1 behaves on an event basis is useful in obtaining a more complete picture for copper leaching. An in depth study of CM1 effluent on an event basis is discussed in sections 6.3 and 6.4.

For the purposes of this research, a “breakthrough” is classified as the time or loading at which media no longer consistently reduces pollutant concentration. A breakthrough is identified on a cumulative mass loading curve as when the gradient of the curve equals or exceeds 1:1 (shown as 0% removal in Figure 24), indicating effluent mass is no longer increasing slowly, but at a rate at least equal to the input. Over 15 years’ worth of loading, none of the sand based mixes showed signs of copper breakthrough. For CM1, breakthrough occurred immediately during the initial dosing. CM2 demonstrated initial leaching of copper, but subsequent removal over the 15 year simulation.

For zinc loading over 15 years (Figure 22b), mass reductions were estimated to be around 72% for sand based mixes. The two commercial media were estimated to reduce zinc mass load by about 50%. None of the media showed signs of breakthrough over 15 years of mass loading.

A phosphorus cumulative loading curve was not generated because the first 6 WQV doses were not sampled.

Overall, there is significant copper and zinc mass removal potential from all media excluding CM1. All three sand based mixes reduced copper and zinc loads more than the commercial mixes. There is little difference in performance between the sand based mixes. (Copper: ~60% removal for sands, 36% and -15% removal for CM2 and CM1 respectively. Zinc: ~70% removal for sands, ~50% removal for commercial mixes).

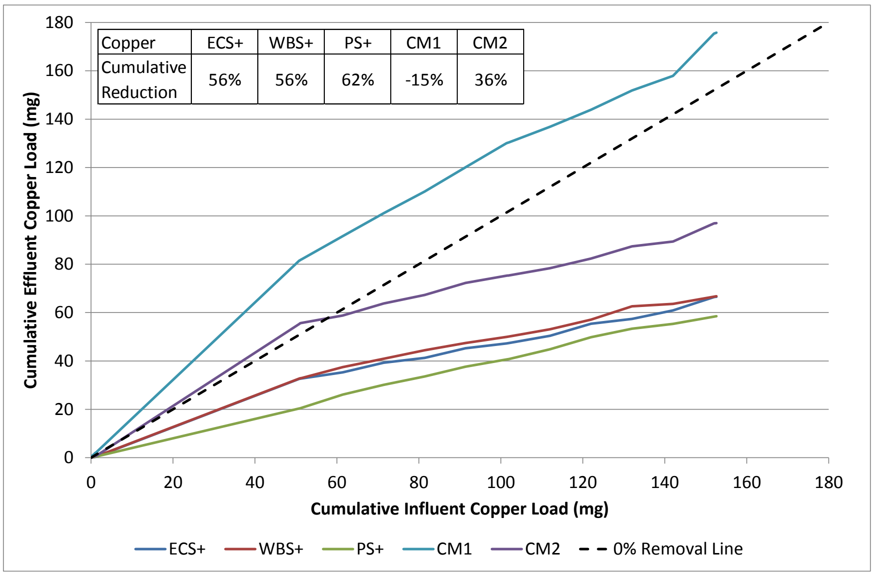
The higher heavy metal removal rate displayed by sand based media compared to the commercial mixes was somewhat unexpected. The commercial mixes contained a much higher proportion of compost, which implies higher organic matter content. Organic matter has a high CEC, and is expected to be beneficial in removing a wide range of heavy metals (Clark & Pitt 2011). CEC of commercial media greatly surpassed the CEC of sand based media (CM1 = 32, CM2 = 25, compared to ECS = 1.1, WBS = 1.8, PS = 6.3, Section 5.6). By association, this means the commercial mixes with a high proportion of compost should remove more heavy metals than sand based mixes with low organic material, however the opposite was measured.

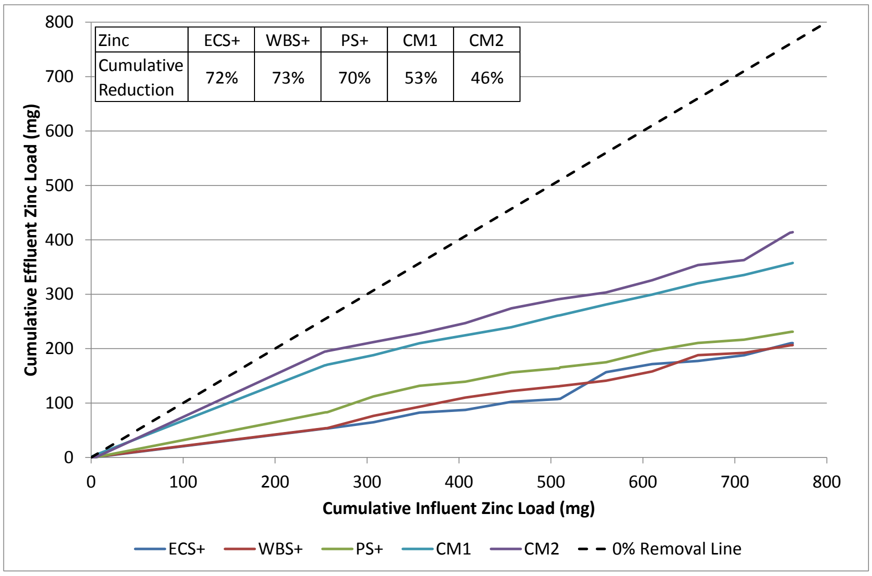
Two factors may have influenced this result; firstly, that both commercial media were pre-loaded with contaminants, so their capacity to remove further contaminants was reduced; and, secondly the commercial media may have been unstable, releasing dissolved organic compounds that mobilised metals.

In the first case, elevated levels of heavy metals are possibly due to contamination in the handling and processing stages of creating a compost (for example, contact with machinery residues, metal storage/sheds), or as part of the compost constituents (for example, copper fungicides and pesticides may be used on plants before they become the green waste that is processed into compost). The New Zealand Standard 4454: 2005 "Composts, Soil Conditioners, and Mulches" provides a voluntary standard for compost production to minimise the potential for "these products to present a risk to the environment or public health" (Standards New Zealand 2005) however, this is not the same as ensuring suitability for use as a treatment medium. The standard suggests copper at 300 mg kg^{-1} and zinc at 600 mg kg^{-1}. If heavy metal contamination of compost were the case, sand based media with relatively low levels of compost would have the advantage of containing less heavy metals. This could mean better heavy metal removal performance for sand based media, compared to the commercial media with a high proportion of compost.

The other possible explanation is organic matter is likely to produce DOC. DOC enhances the mobility of copper in soil by acting as a colloidal transport (Altaher 2001). The same process is also exhibited on zinc, but to a lesser extent (Christensen et al. 1996). Since commercial mixes contain more organic material, it is likely there is more DOC and hence greater mobility of copper and zinc when compared to sand based media.

Figure 24a-b Cumulative copper and zinc mass loadings





6.3 Event-based pollutant removal efficiency

Figure 25-27 show influent versus effluent concentrations for 15 WQV doses. Any points below the solid black gradient line indicate removal of pollutant, while any points above the line indicate leaching. The variation in influent concentrations are due to a combination of improving synthetic stormwater mixing methodology (see 4.7.3.2) and discrepancy from working at trace levels.

Figure 25a-e shows the 15 effluent concentrations of copper corresponding to influent dosing concentrations. There is almost complete removal of copper in ECS+, WBS+, PS+, and CM2 columns. Copper removal in CM1 is less clear, with leaching occurring in the first 5 out of 15 WQV doses, suggesting CM1 started with an elevated level of readily-mobilised copper in the media. Following the initial leaching, CM1 showed significant copper retention for 7 out of the remaining 10 WQV doses, however levels of copper in those 7 WQV effluents (7 μg/L) were significantly higher than the other four media columns (3 μg/L), indicating poorer copper removal (30% removal for CM1 compared to 70% removal for other media). The other three CM1 WQV doses exhibited no change between influent and effluent concentrations.

Overall, four out of five media columns showed excellent and consistent levels of copper removal for WQV storms. For the four media, WQV effluent copper concentrations had a median of 3 μg/L and average absolute deviation ranging from 0.7 μg/L for ECS+ to 2.9 μg/L for CM2. This corresponds to 75%, 70%, 65%, and 70% average event-based copper removal for ECS+, WBS+, PS+, CM2, respectively.

Figure 26a-e similarly shows removal or leaching of zinc for 15 WQV doses. Similarly to copper, ECS+, WBS+, PS+, and CM2 display almost complete removal of zinc from influent doses. Five occurrences of zinc leaching in the CM1 column again they took place in the first five WQV doses, indicating CM1 initially had an elevated level of readily-mobilised zinc in the media. Following the initial leaching, CM1 showed zinc retention for 8 out of the remaining 10 WQV doses, however the levels of zinc in the effluents (20 μg/L) showing removal were significantly higher than effluent from the other four media columns (10 μg/L), indicating poorer zinc removal (60% removal for CM1, compared to 80% removal for other media). The other two CM1 doses exhibited no change between influent and effluent concentrations.

Overall, four out of five media columns showed excellent and consistent levels of zinc removal for WQV storms. For the four media, WQV effluent zinc concentrations had a median of 8 μg/L and average absolute deviation ranging from 2.6 μg/L for ECS+ to 9.5 μg/L for PS. This corresponds to 87%, 82%, 88%, and 77% average event-based zinc removal for ECS+, WBS, PS, CM2 respectively.

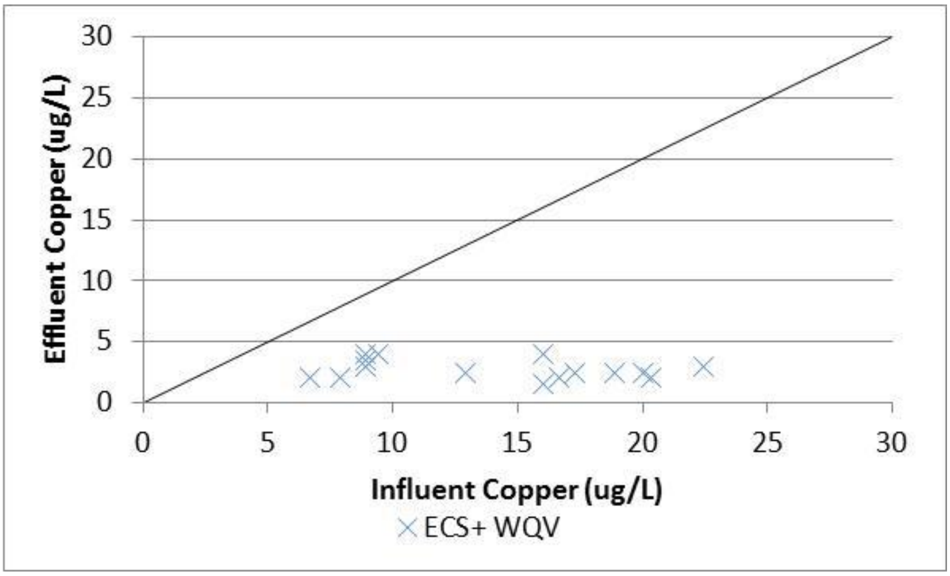
Bioretention cells using any of the sand based filter media should expect similar dissolved copper and dissolved zinc removal rates on an event-basis. CM2 is also similar to sand based media in terms of dissolved zinc removal. The percentage removals for copper and zinc do not match the over 90% and over 95% removals found by Hatt et al. (2008) and Davis et al. (2003) respectively. The two studies have measured total concentrations, which include particulate heavy metals. No particulate copper or zinc was dosed herein, hence the high concentrations of heavy metals attached to sediments (which are relatively easily removed via filtration) were not accounted for. The significant removal of dissolved heavy metals to produce effluent with only trace concentrations is an important result with potential practical ecosystem benefits if implemented.

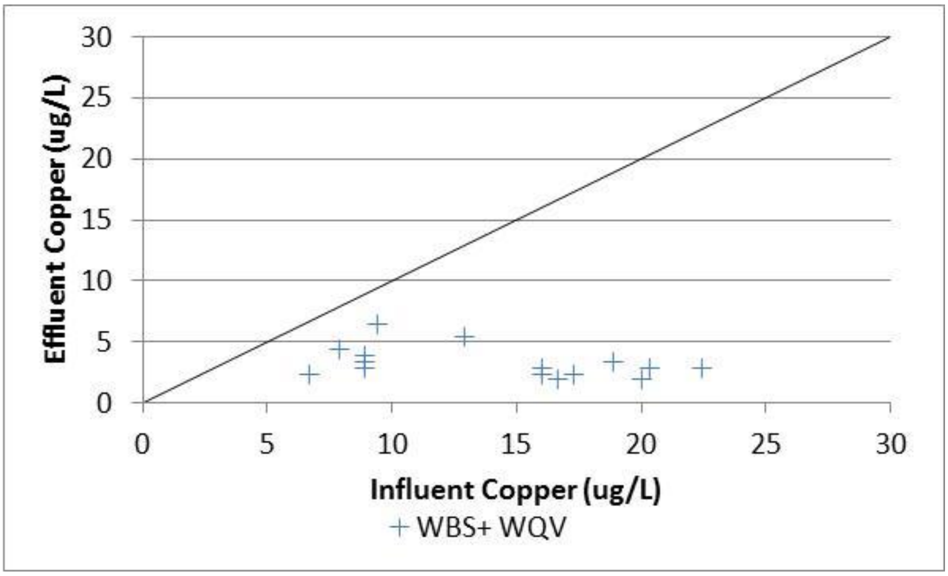
Figure 27a-e shows leaching of phosphorus for nine WQV doses (WQV 7 through 15, data are not available for the first six doses). All five media columns showed high levels of phosphorus leaching, which is consistent with the predictions based on media chemistry (Section 5.6). CM2 displayed the highest amount of phosphorus leaching, consistently producing effluent in the 2,500 to 3,500 μg/L range from an influent concentration of 65 μg/L. CM1 was the "best" performing, with effluent phosphorus concentrations consistently limited to around 500 μg/L. ECS+, WBS+, and PS+ effluent phosphorus concentrations are all consistently in a similar range (500 to 2,000 μg/L); higher than CM1, but lower than CM2. Overall, all media increased phosphorus concentrations from influent to effluent by two or up to three orders of magnitude.

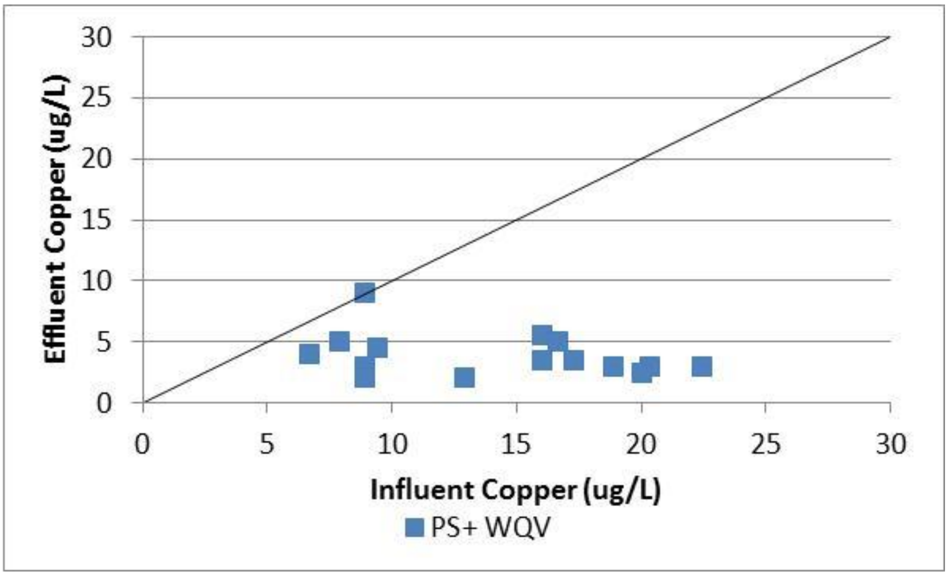
Studies have shown the presence of organic matter to be associated with phosphorus leaching (Bratieres et al. 2008; Peltier & Carbone 2011). CM2 and CM1 are the two media with the highest organic contents, however they display opposite behaviour in terms of phosphorus leaching. CM1 leaching was even lower than the sand based media, which contained relatively low amounts of organic matter. Leaching potential was discussed in Section 5.6. Varied levels of leaching are attributed to varied media chemistry.

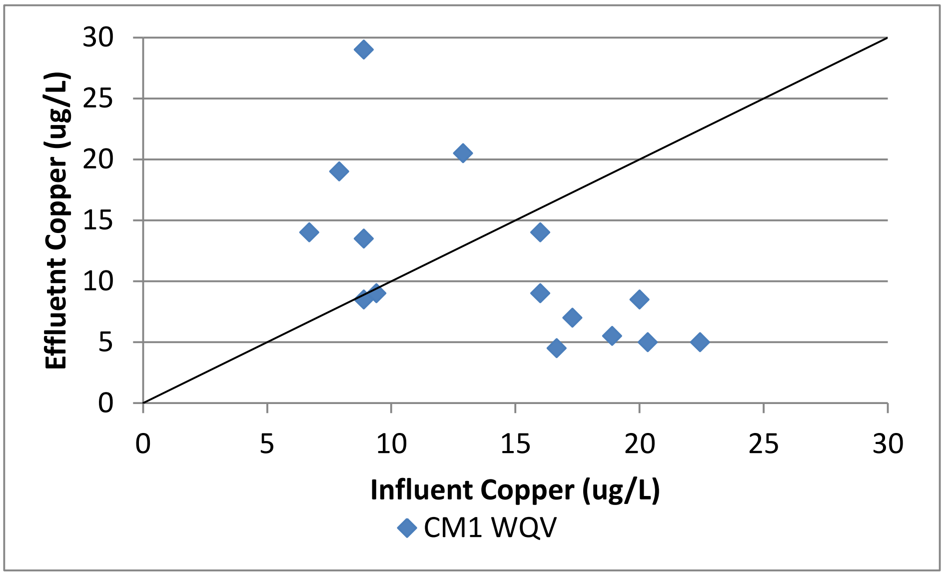
Results from the 5YCONC dose (Figure 28) suggest phosphorus was being removed by the media after six WQV and one 5YCONC dose, meaning media were not saturated with phosphorus initially. However phosphorus removal from the 5YCONC result does not indicate the WQV doses that preceded it also removed phosphorus. Media sorption can have different responses when exposed to high concentrations (5YCONC, 1YCONC) and low concentrations (WQV) of phosphorus. High concentrations of phosphorus may cause other ions to be displaced from media anion sites, to make anion sites available for phosphorus ions. For high concentrations of phosphorus, the ion usually displaced from the anion sites is the OH^- ion, followed by Cl^-, NO^{3-} and SO_4^{2-} (Parfitt 2011). The displaced ions may not have been displaced if instead it was a low concentration of phosphorus dosed. The theory of different media responses for high concentration and low concentration phosphorus doses is supported by Figure 29a-e. The figure shows four out of five media display phosphorus removal when exposed 1YCONC doses (high concentration). However, the WQV (low concentration) doses which are interspersed between the 1YCONC instead display leaching. These results could be understating the potential for P leaching.

Figure 25a-e Copper WQV dosing






Effluent Copper (ug/L)
Influent Copper (ug/L)
PS+ WQV


Effluetnt Copper (ug/L)
Influent Copper (ug/L)
CM1 WQV

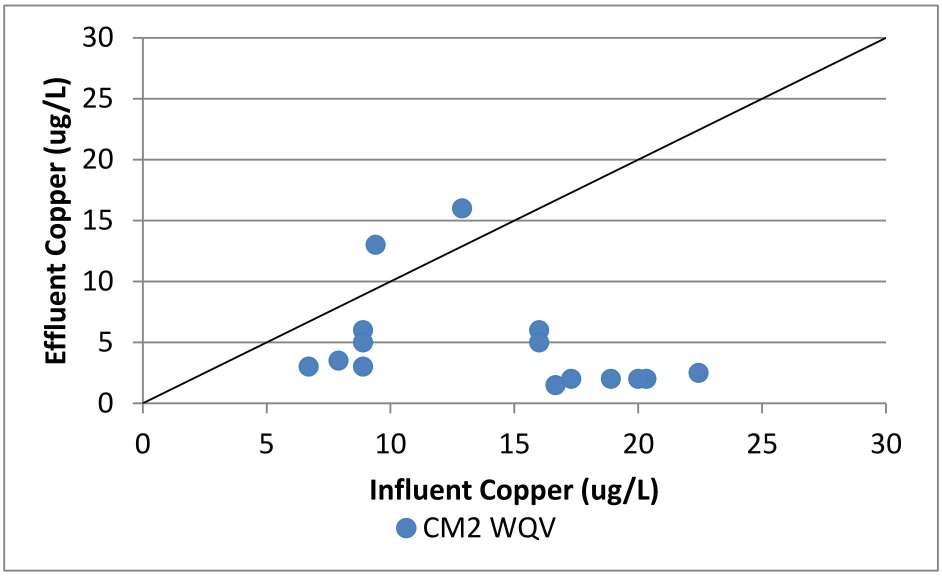
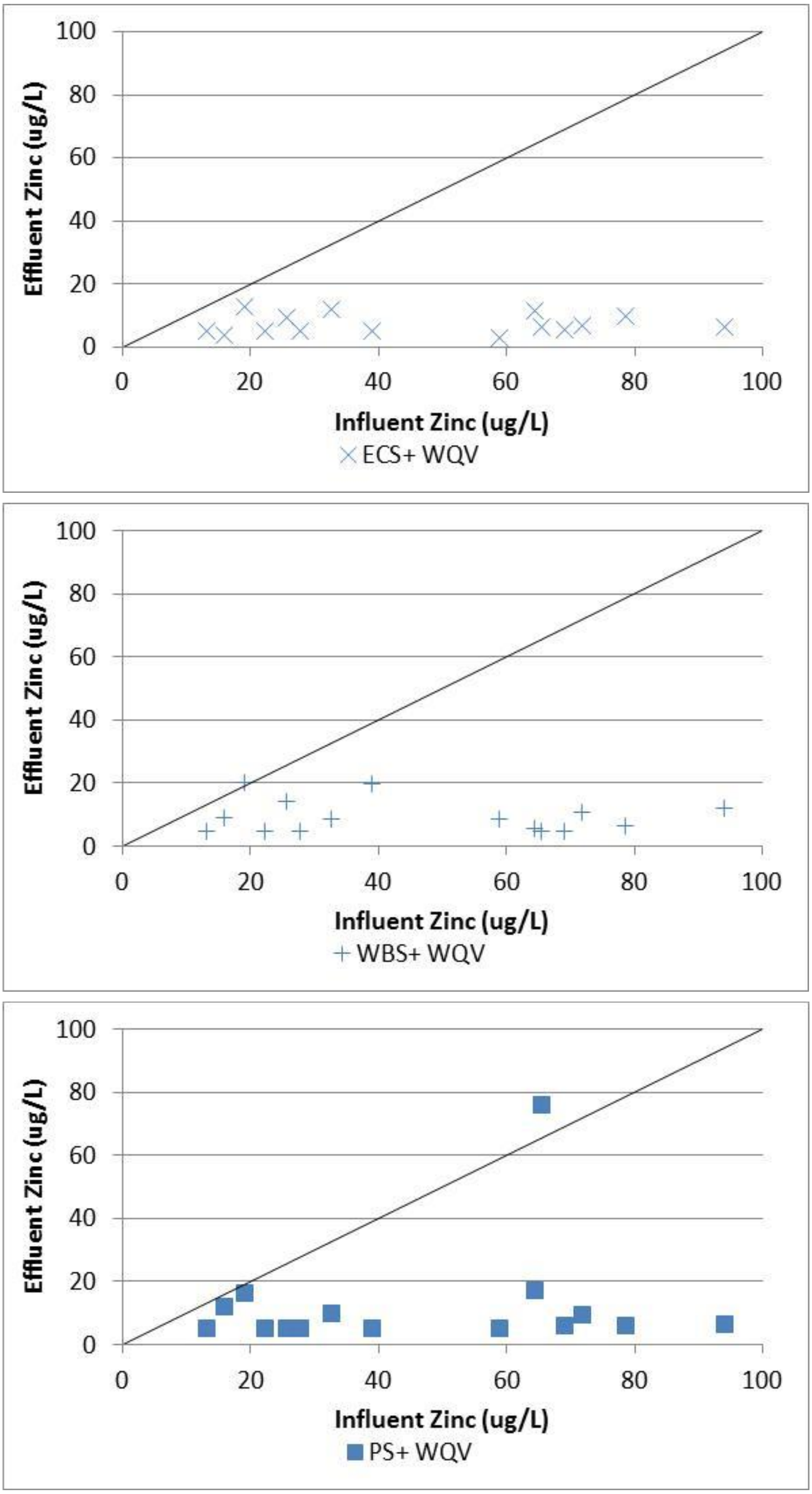
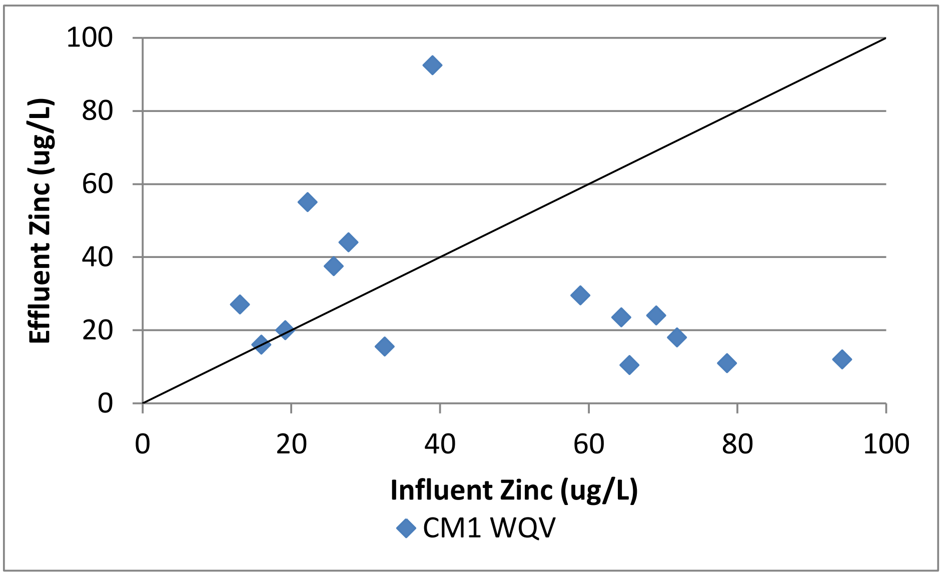

Effluent Copper (ug/L)
Influent Copper (ug/L)
CM2 WQV

Figure 26a-e Zinc WQV dosing




Effluent Zinc (ug/L)
100
80
60
40
20
0
0
20
40
60
80
100
Influent Zinc (ug/L)
CM1 WQV

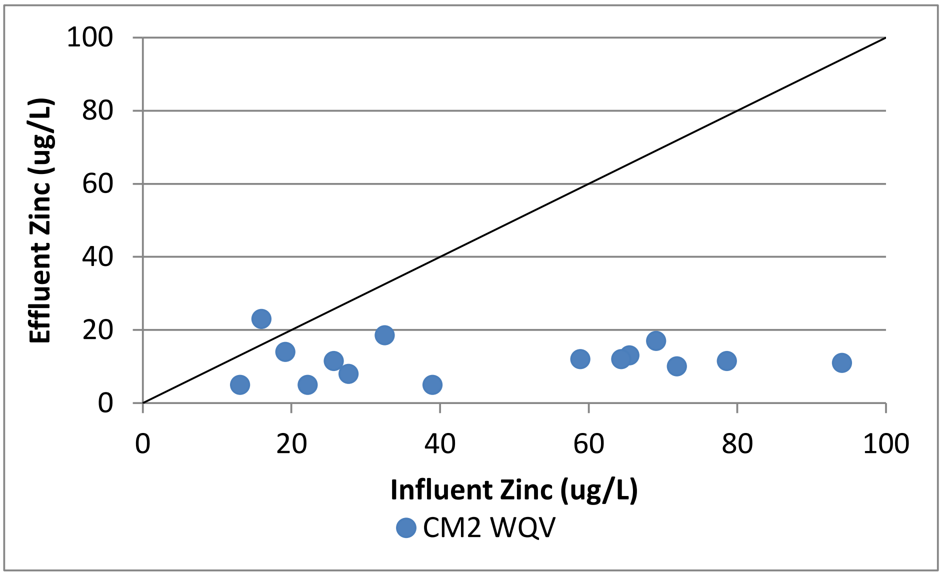
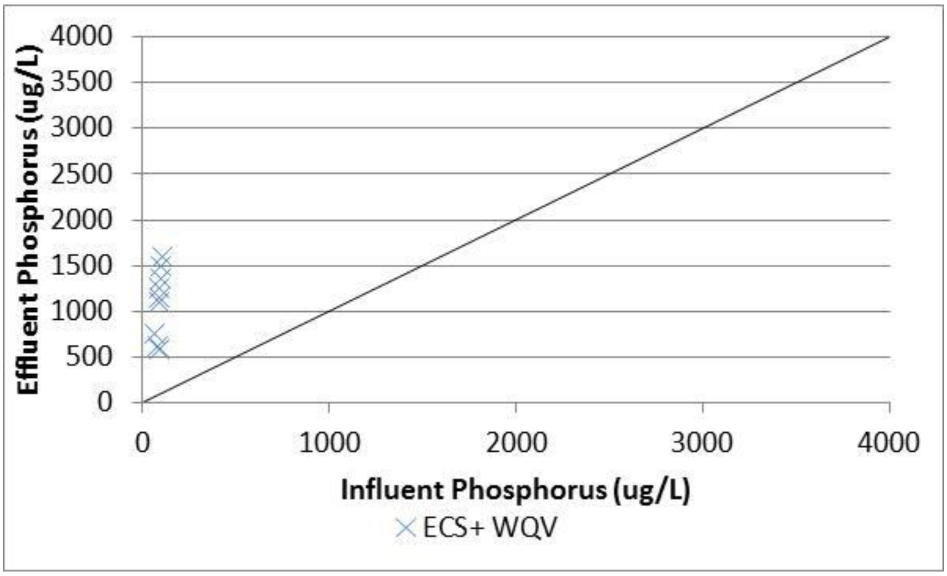
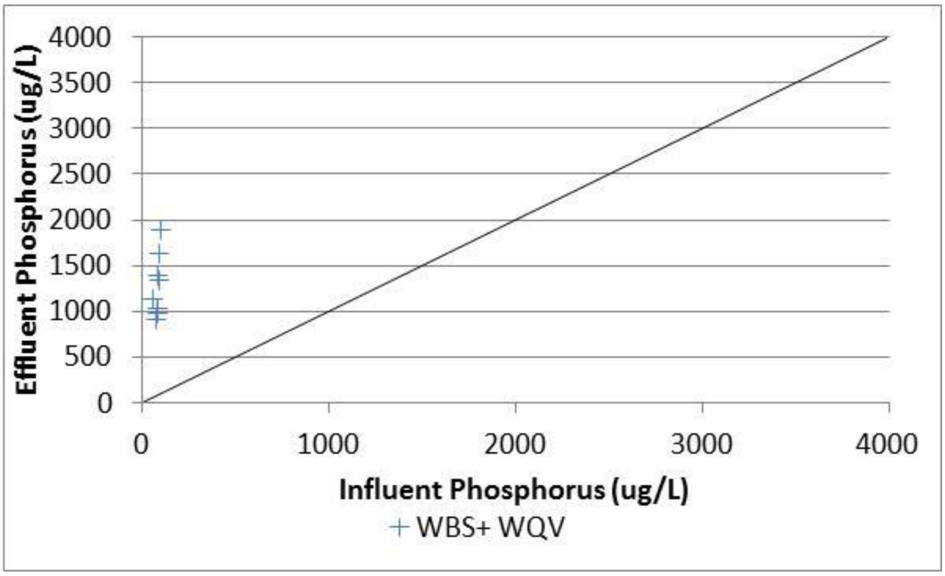
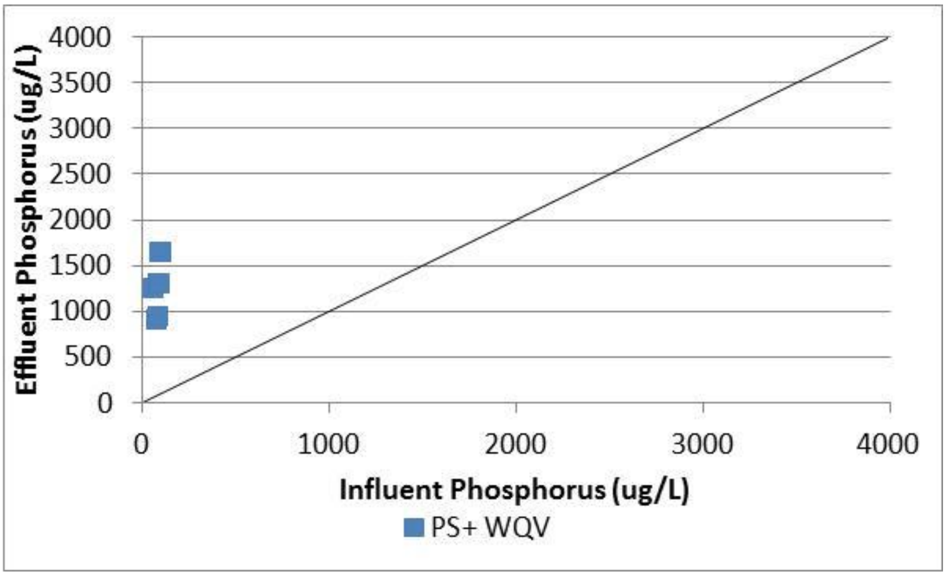
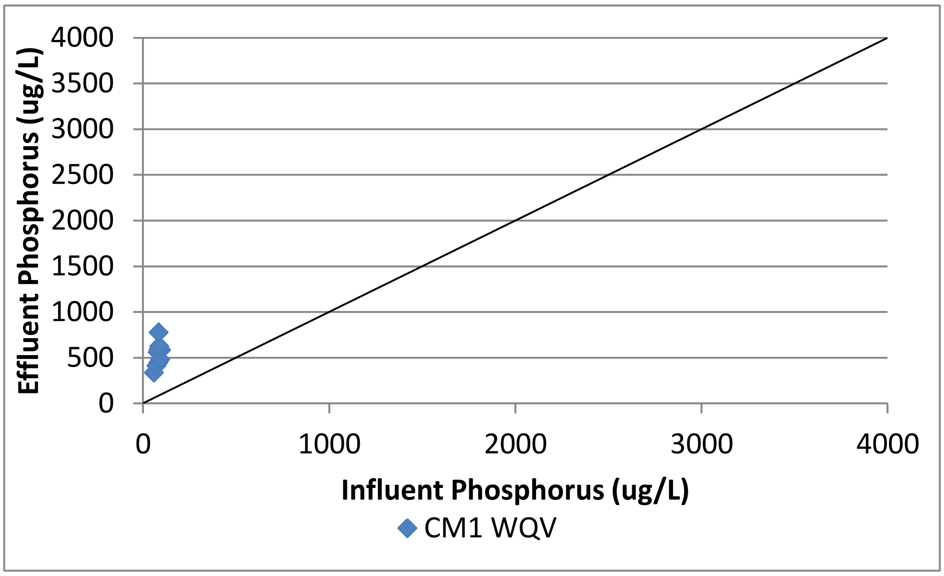

Effluent Zinc (ug/L)
100
80
60
40
20
0
0
20
40
60
80
100
Influent Zinc (ug/L)
CM2 WQV

Figure 27a-e Phosphorus WQV dosing









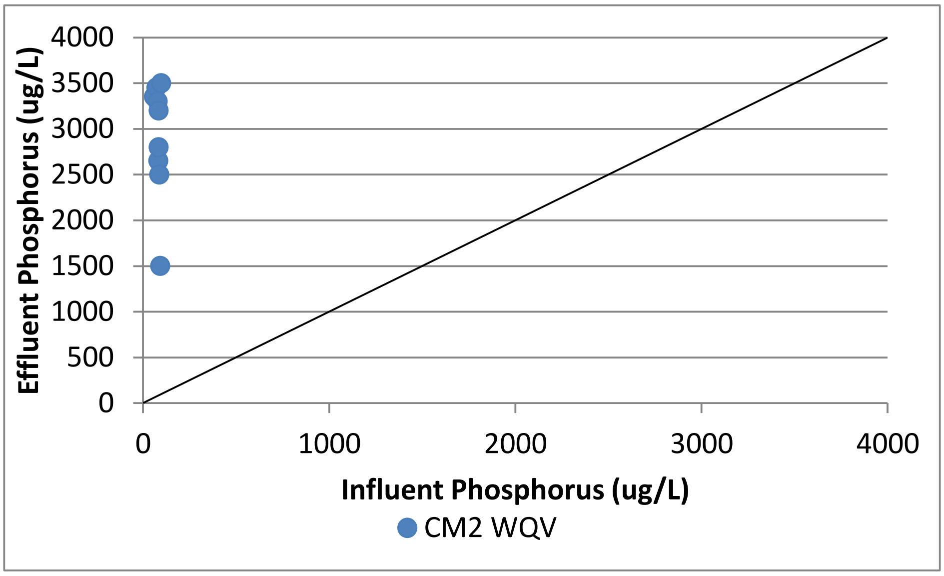


Figure 28 Phosphorus 5YCONC dosing

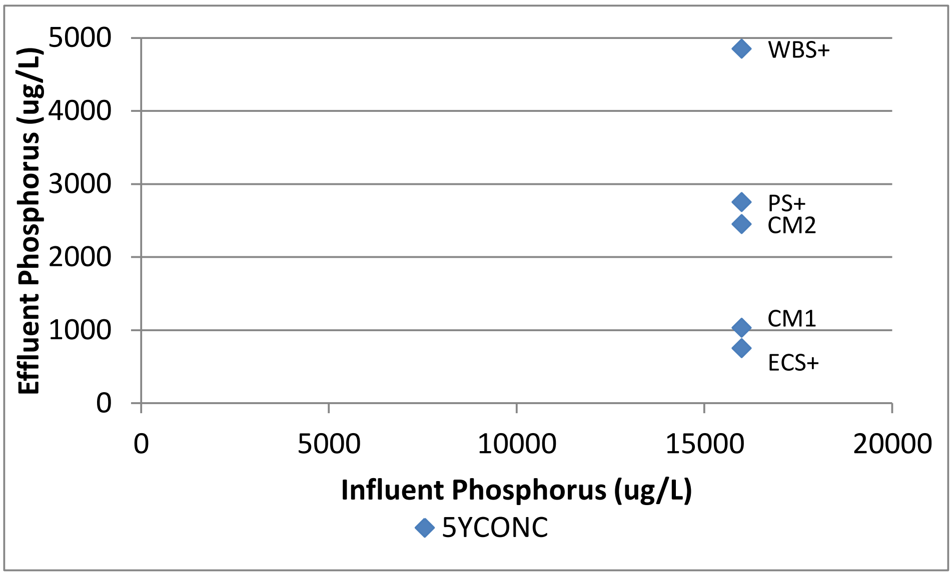
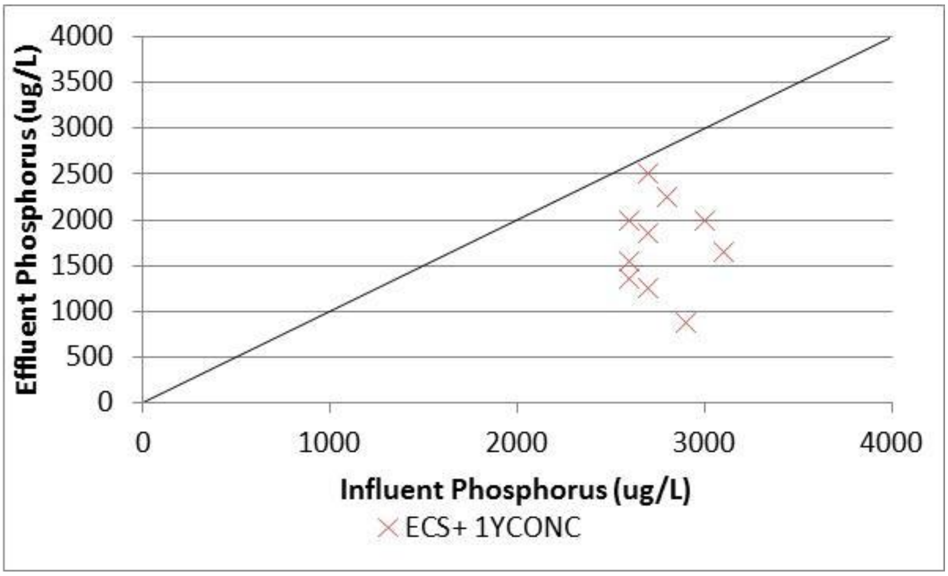
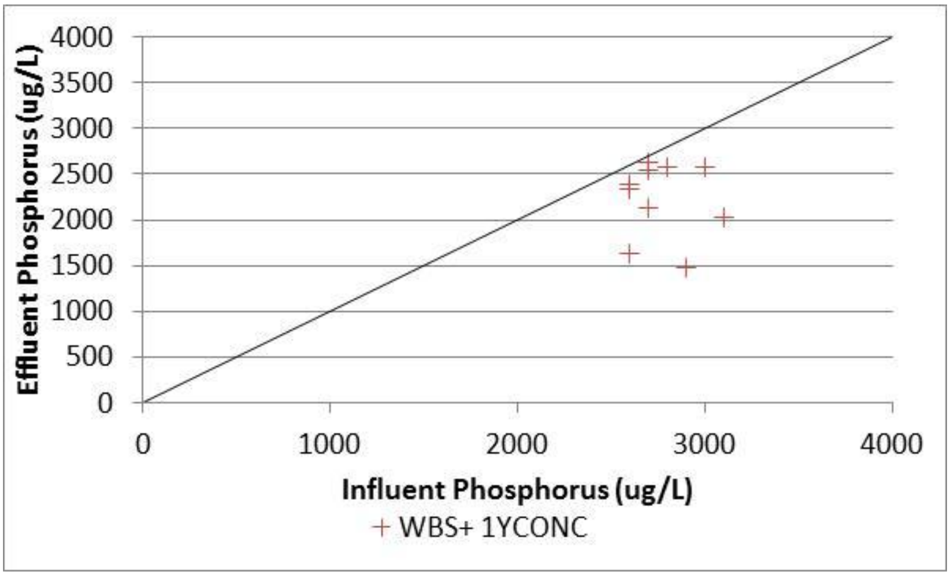
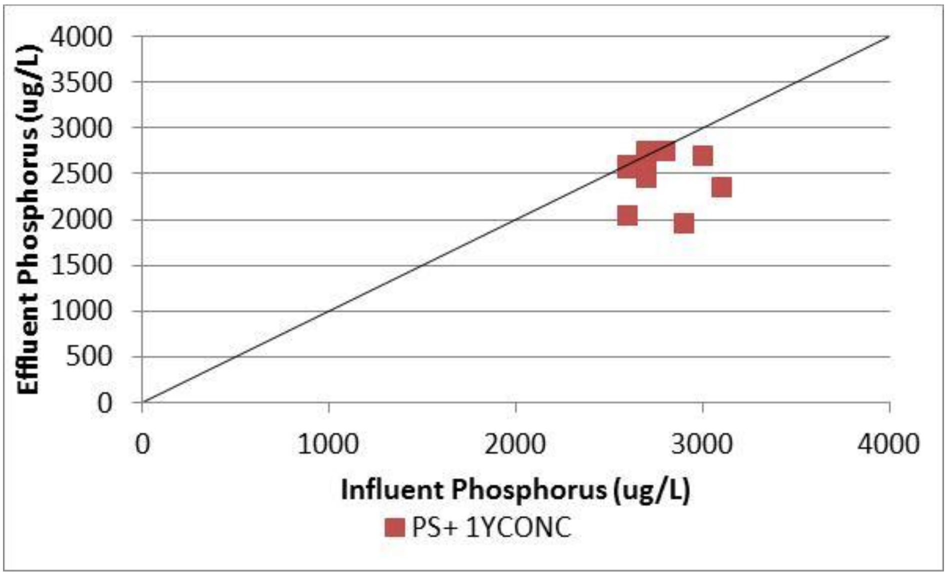
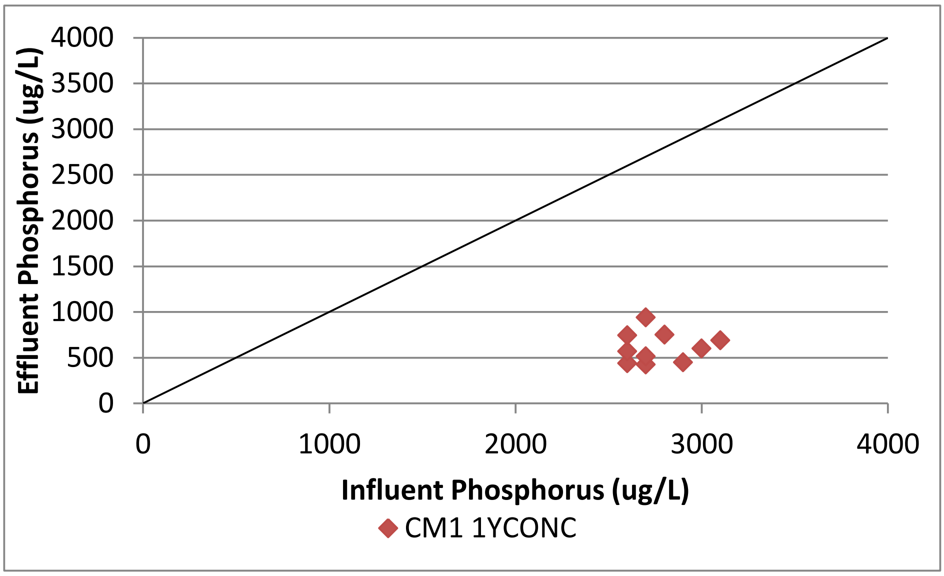


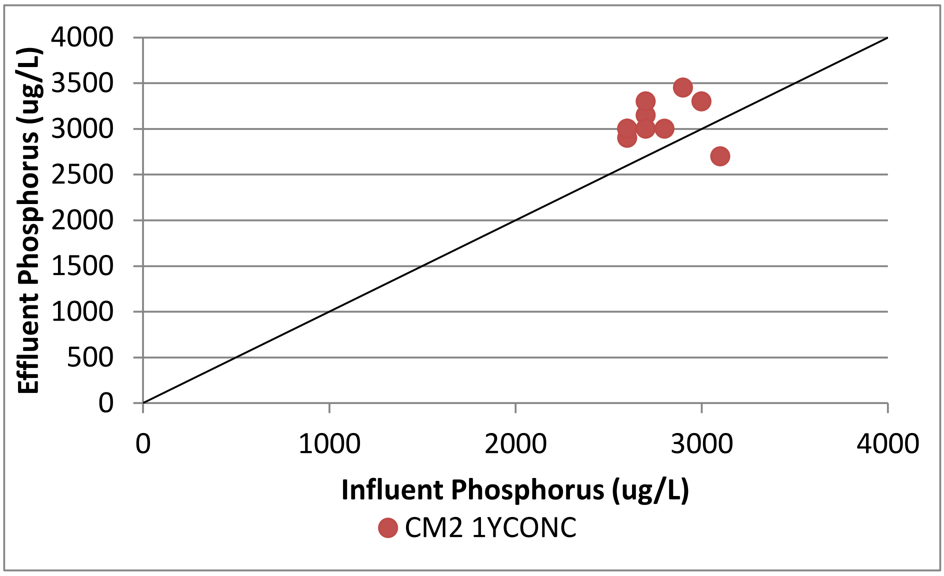
Figure 29a-e Phosphorus 1YCONC dosing











6.4 Effect of media aging on pollutant removal efficiency

Figure 30 shows effluent copper concentrations from media columns after WQV doses at different stages of aging. ECS+, WBS+, and PS+ media columns consistently produce effluent copper concentrations below 5 μg/L across 15 years of media aging, indicating cleaner effluent and consistent performance. CM2 was similarly below 5 μg/L except at 5 years of aging, where the mean was closer to 10 μg/L and there was significant variability of the particular set of data. Higher levels of organic matter (as found in CM2 compared to the sand based mixes) are directly related to production of dissolved organic carbon, which acts to increase copper mobility (Altaher 2001).

CM1 had elevated effluent copper concentrations (19 μg/L) at 0 years aging; higher than the influent copper concentration (10 μg/L). However, effluent copper concentrations from CM1 systematically reduced to less than influent concentrations as the media was aged, indicating eventual copper removal potential (50% removal at 10 years of aging). The initial leaching of copper is indicative of a “first flush” of copper initially residing within the media. This means CEC sites in CM1 were initially saturated with copper ions, and additional copper concentrations found in the effluent originated from the media itself. Soil chemistry tests show CM1 media has a base CEC saturation of 140% (Section 5.6), supporting this theory. Although removing the first flush effect was part of the methodology (section 4.7.1), the data suggest that CM1 required a significantly longer flushing period before consistency could be achieved. The pollutant mass released during a first flush after installation will be insignificant compared to the quantity of pollutants removed from stormwater runoff throughout the life of the bioretention cell. In practice, it should not take multiple years of stormwater to get past the first flush phase. For this investigation, aging was simulated with small volumes of stormwater. These small volumes were insufficient in getting past the first flush, while in realistic situations the first flush would be over after several full volume storms with insignificant aging. If this first flush is released in high concentrations to surface waters in low flow conditions the impact on aquatic life could be devastating; such conditions have occurred in Auckland where bioretention cells are constructed in summer and subsequently irrigated.

Figure 31 shows effluent zinc concentrations after WQV doses at different stages of aging. ECS+, WBS+, PS+ and CM2 media columns typically produced consistent zinc removal across all 15 years of media aging. The mean effluent zinc concentration for these media was 10 μg/L. CM1 initially had an 8% removal rate of zinc, however similarly to CM1 behaviour with copper, the removal efficiency of zinc increased with 5 and 10 years aging (59% removal and 73% removal respectively). In-environment trigger concentrations for 95% level of protection of species for freshwater and marine water are 8 μg/L and 15 μg/L zinc respectively (ANZECC 2000). In terms of zinc pollution, effluent from all media except CM1 was cleaner than marine water trigger levels, even without the benefit of dilution in a mixing zone (which is considered in the water quality standard).

Figure 32 shows effluent phosphorus concentrations after WQV doses at different stages of aging. There are no data for 0 years aging. Effluent phosphorus for every media is often several orders of magnitude higher than the influent concentration (65 μg/L), which was discussed in sections 5.6 and 6.3. In addition, as discussed in section 4.7.3.2, influent phosphorus concentrations are one third of concentrations found in field studies of motorway stormwater runoff (Section 4.7.3.2). The consequences of high phosphorus leaching can be severe for slow-moving, phosphorus-limited, freshwater aquatic environments, with a particular concern for eutrophication.

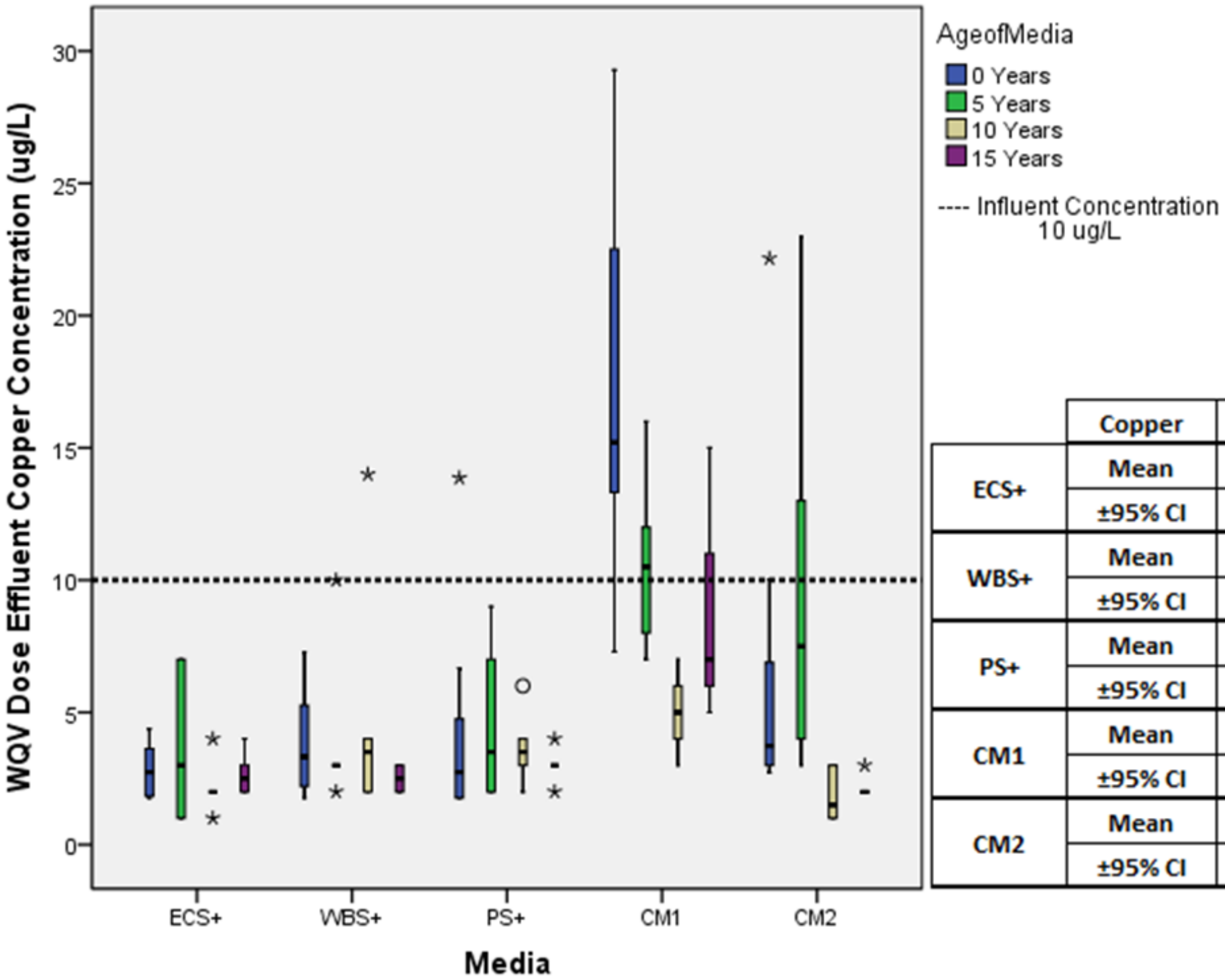
At 5 years, 10 years, and 15 years (Figure 32), CM2 has a higher effluent phosphorus concentration than the other media (approximately 3,200 μg/L for 5 to 10 year, and 2,300 μg/L for 15 year). The decrease in concentration after 15 years is a sign that the phosphorus in the media is finally in the process of being depleted after multiple leachings.

PS+ effluent phosphorus remains stable across 5, 10 and 15 years of aging (1,200 μg/L). ECS+, and WBS+ have effluent phosphorus concentrations increasing substantially after 10 years aging when compared to after 5 years aging (600 μg/L to 1,300 μg/L for ECS+, 1,000 μg/L to 1,400 μg/L for WBS+). Effluent phosphorus at 15 years is similar as at 10 years. The result suggests a change in ECS+ and WBS+ media occurred after 10 years of phosphorus dosing, prompting greater phosphorus leaching. CM1 was clearly the "best" performing in terms of effluent phosphorus concentrations (500 μg/L). While the level of performance decreased after 10 years aging compared to 5 years aging (400 μg/L to 600 μg/L), the effluent concentrations were still consistently lower than the other media, however still much higher than the influent concentrations.

Plants are able to reduce phosphorus in media by uptake, or export phosphorus if poorly designed or maintained systems experience significant plant die-back. As there were no plants in the laboratory setup, the net export of phosphorus is likely overestimated. However, it is not expected that plant uptake alone would be capable of reducing effluent phosphorus to reasonable levels. Also, because the 15 year timeframe is compressed over a few weeks, the mineralisation of organic matter that would have occurred over 15 years (releasing phosphorus) is not measured. If the mineralisation rate is greater than plant uptake, the media could release additional phosphorus.

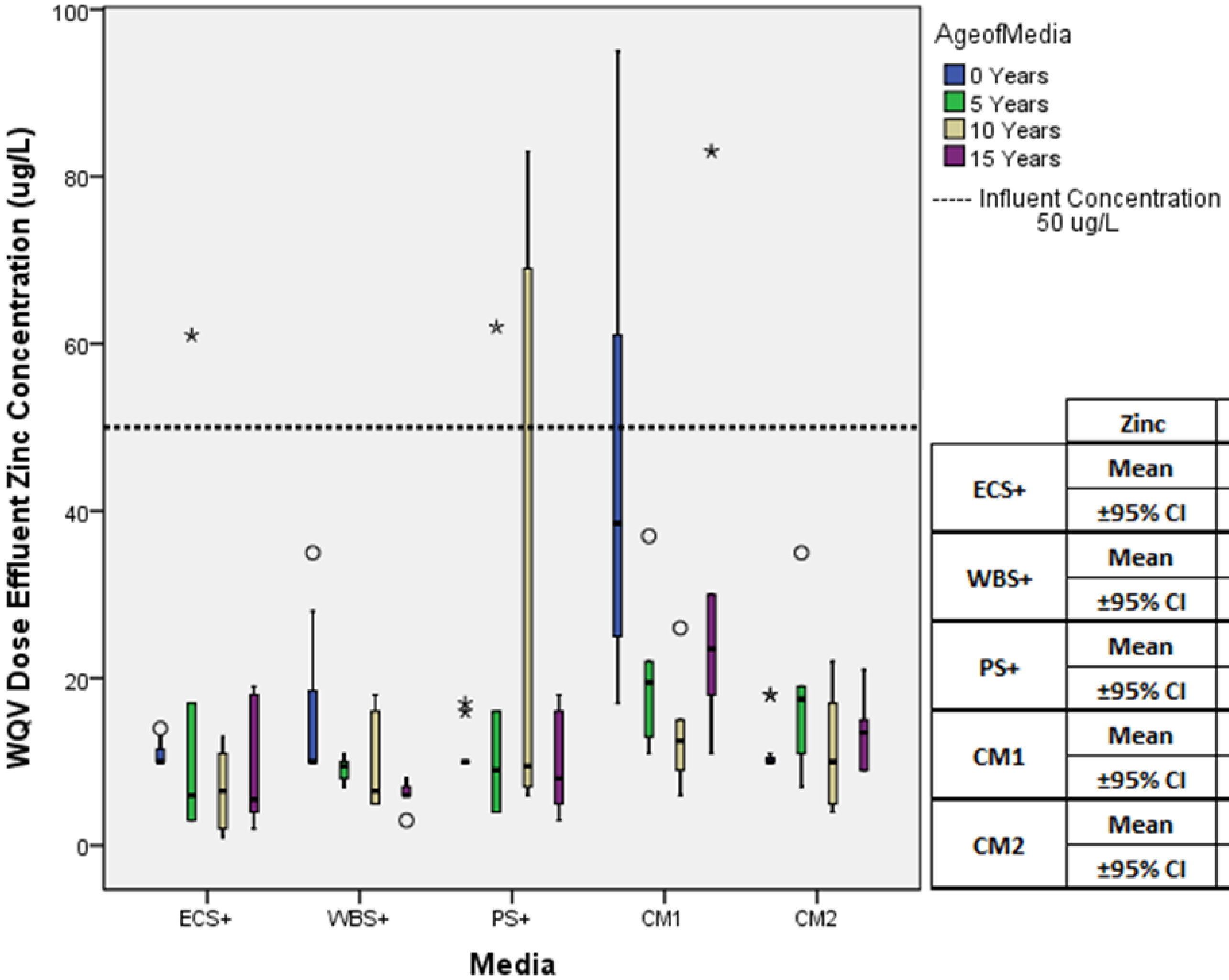
The extreme leaching of phosphorus raises concerns on whether any of the media investigated are fit for bioretention filter media purpose in phosphorus-sensitive receiving environments without further additives or amendments. A limited reporting of possible media additives from a literature review are discussed in section 3.2, however testing is required to establish the quality and suitability of local additives.

Figure 30 Effluent copper concentrations at different media ages



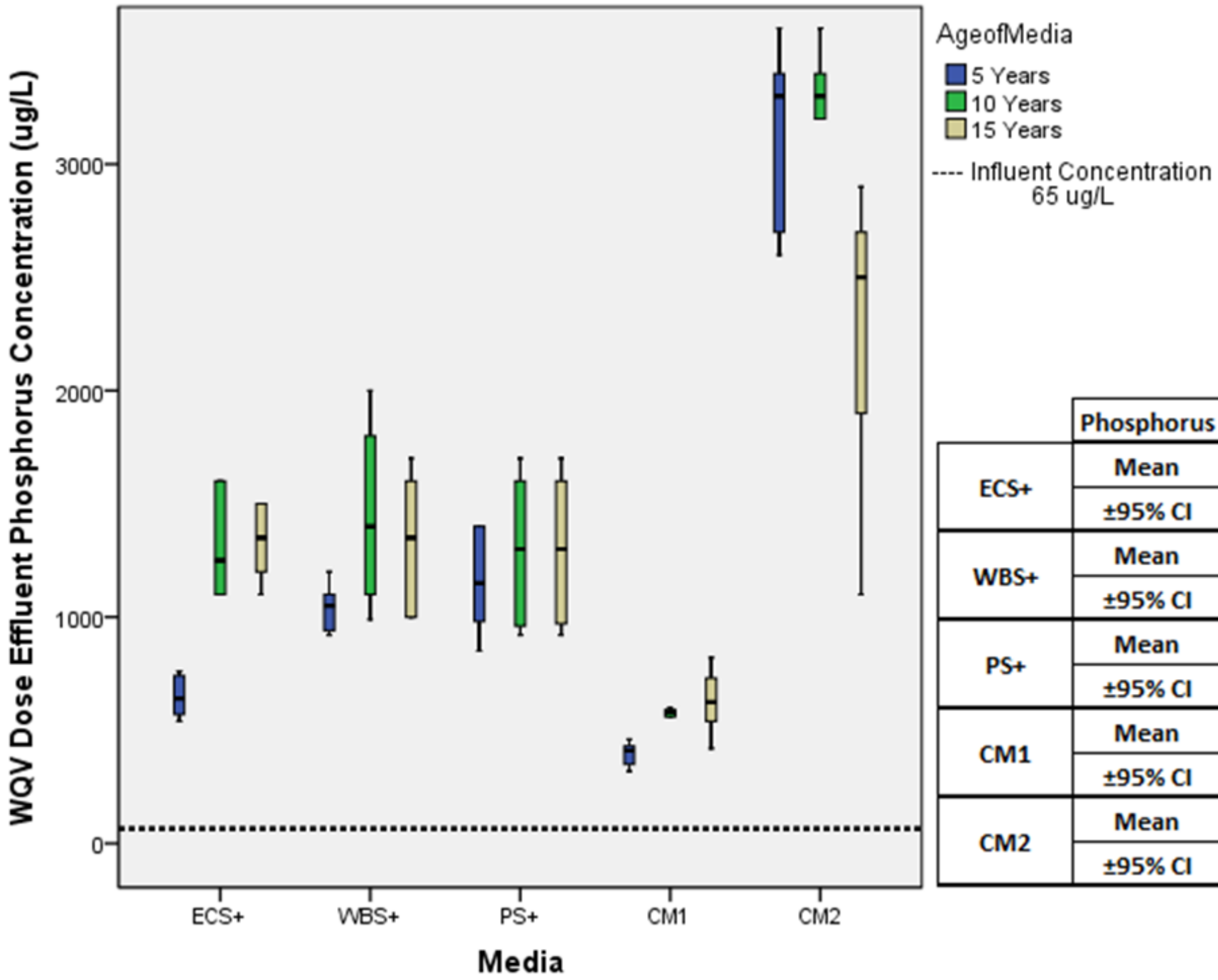
	Copper	0 Year	5 Year	10 Year	15 Year
ECS+	**Mean**	2.8	11.3	2.2	2.7
	±95% CI	0.6	21.6	1.0	0.9
WBS+	**Mean**	3.8	4.0	4.8	2.5
	±95% CI	1.2	3.1	4.8	0.6
PS+	**Mean**	4.0	4.5	3.7	3.0
	±95% CI	2.2	3.0	1.4	0.7
CM1	**Mean**	18.9	10.7	5.0	8.5
	±95% CI	6.6	3.4	1.6	4.0
CM2	**Mean**	6.0	9.7	1.8	2.2
	±95% CI	3.5	7.9	1.0	0.4

Figure 31 Effluent zinc concentrations at different media ages



	Zinc	0 Year	5 Year	10 Year	15 Year
ECS+	Mean	10.8	16	6.7	9
	±95% CI	1	23.8	5.3	7.9
WBS+	Mean	15	9.2	9.5	6
	±95% CI	5.6	1.6	6.2	1.8
PS+	Mean	11.1	17.3	30.7	9.7
	±95% CI	1.6	23.5	37.2	6.4
CM1	Mean	46	20.3	13.5	31.5
	±95% CI	16.7	9.7	7.4	27.5
CM2	Mean	11.4	17.8	11.3	13.5
	±95% CI	2	10.1	8	4.7

Figure 32 Effluent phosphorus concentrations at different media ages



	Phosphorus	0 Year	5 Year	10 Year	15 Year
ECS+	Mean	NoData	648.3	1316.7	1333.3
	±95% CI		93.5	243.1	171.4
WBS+	Mean	NoData	1043.3	1448.3	1333.3
	±95% CI		113.7	410.2	308.9
PS+	Mean	NoData	1155	1296.7	1298.3
	±95% CI		234.1	335.1	333
CM1	Mean	NoData	396.7	578.3	626.7
	±95% CI		54.6	16.8	157.6
CM2	Mean	NoData	3150	3333.3	2266.7
	±95% CI		423.7	158	695.1

6.5 Effect of drying period

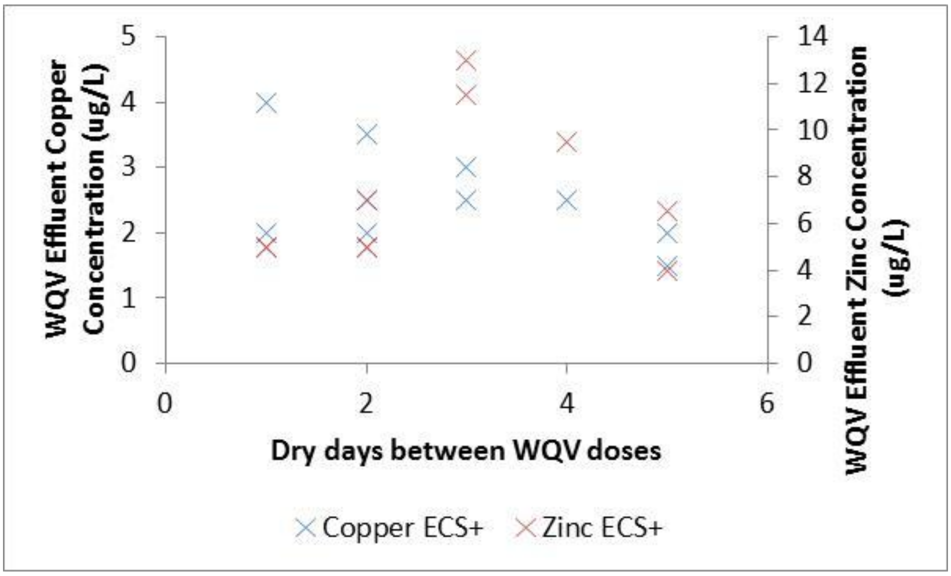
The effect of drying periods between doses on water quality was investigated (Figure 33a-e). In laboratory testing, dosing was performed on a tightly scheduled basis. Real storm events are often variable in frequency of occurrence, intensity, volume, and duration. The unpredictable storm events cause intermittent wetting and dry periods for bioretention filter media. Wetting and drying of soils can significantly affect the chemical state and structural state of the soil, as well as plant activity. These changes will in turn affect the chemical and physical interaction between soil and pollutant. Blecken et al. (2009) found antecedent drying exceeding 3 - 4 weeks could cause significantly worse performance in bioretention cells.

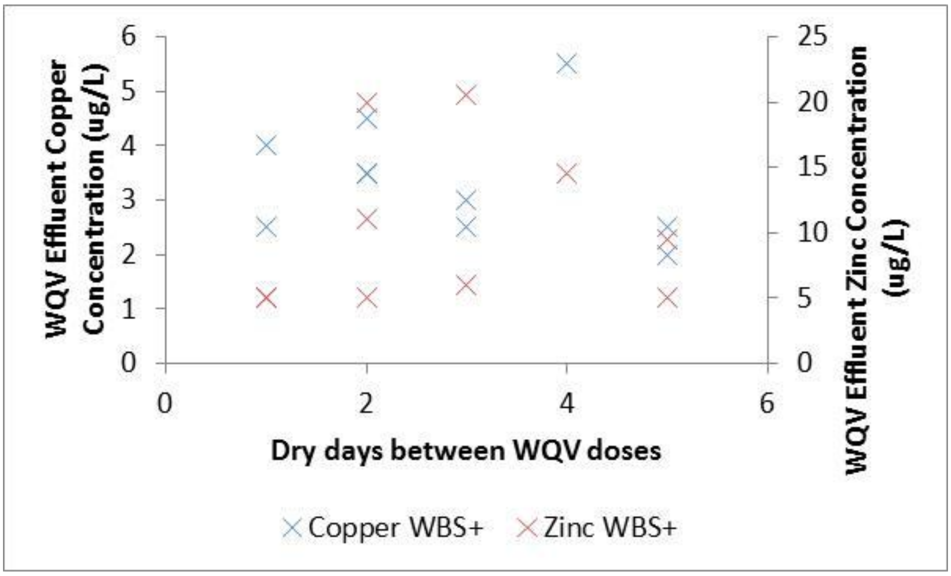
In Auckland, there are typically 137 wet days (> 1.0 mm rainfall) per year (NIWA Science 2000). Discounting seasonal variations, this is equivalent to 2.6 wet days per week. It is unlikely for Auckland to experience regular 3 - 4 week dry periods in which bioretention cells would deteriorate in performance.

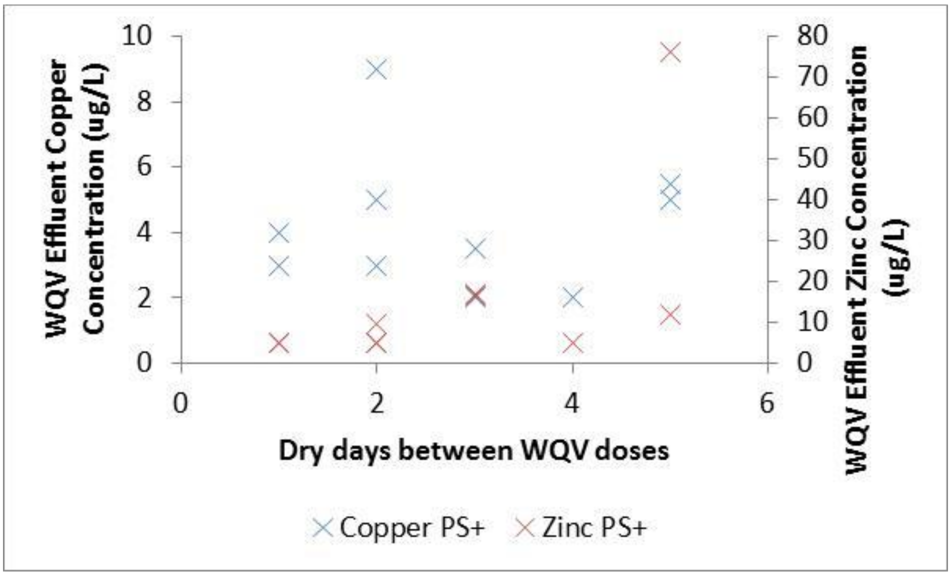
The results of this investigation show no evidence of drying periods up to 5 days having an effect on effluent water quality for any of the media. This is demonstrated by the lack of trends in Figure 33. The effect of drying periods of greater than 6 days was not investigated due to tight timeframes, however the need for testing extended drying periods is debatable when considering Auckland's rain conditions.

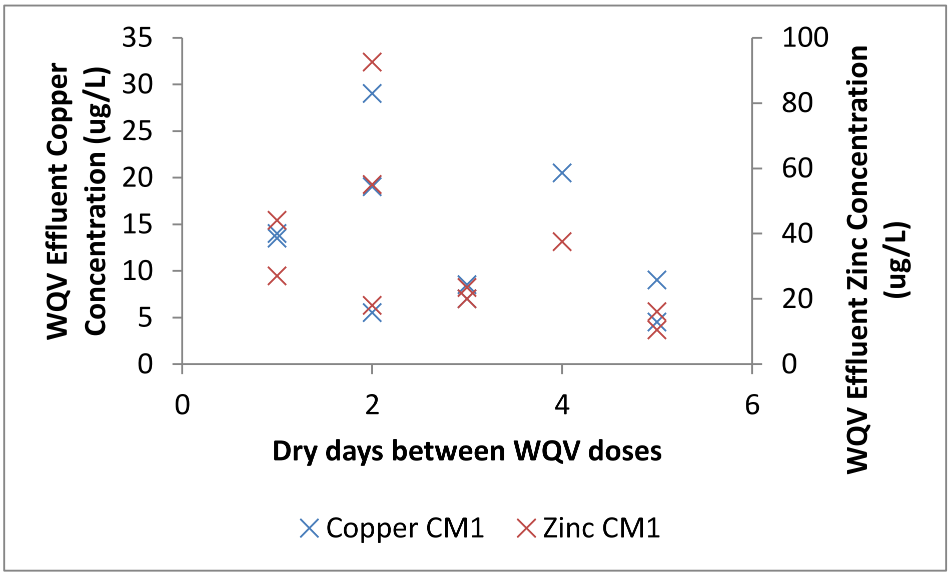
Wetting and drying can also cause substrate structural changes. At the end of the plant pot trial (Section 7), the plants were dried and water with-held for a month; the two sand-based media (Pumice and East Coast Sand) showed no shrinkage. In contrast, CM1 shrank from the edges of the pot. Shrinkage and cracking creates zones of preferential flow allowing stormwater to bypasses the matrix and therefore decreases contaminant removal efficiency.

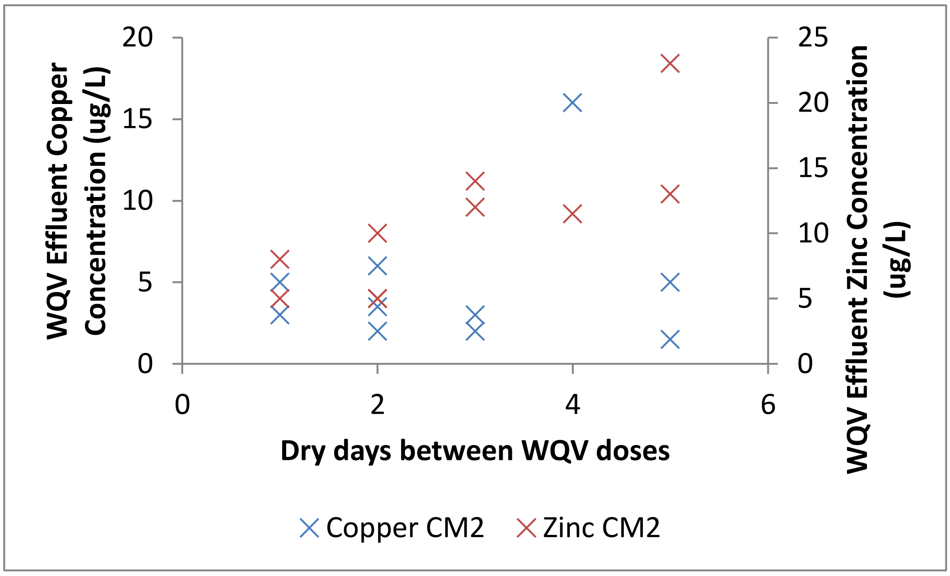
Figure 33a-e WQV effluent concentrations after varying dry periods









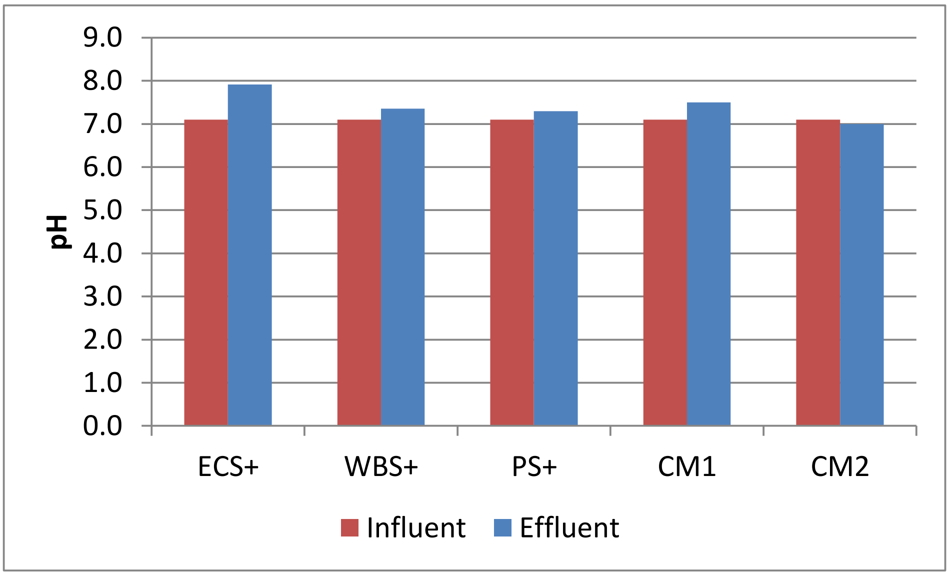


6.6 pH

Influent and effluent pH are shown in Figure 34. The figure is an average of results from WQV7 and 1YCONC2. There were no differences in pH between the WQV and 1YCONC doses measured. Effluent from CM2 pH decreased slightly from the influent, while all other media had increasing effluent pH.

Influent pH was approximately 7.1, which is within the range where copper is most mobile in soils (ph 6.24 to 7.24) (Altaher 2001). Despite these optimal conditions for copper mobility, copper was well retained by adsorption in four out of five columns (section 6.3).

Figure 34 pH


9.0
8.0
7.0
6.0
5.0
4.0
3.0
2.0
1.0
0.0
pH
ECS+
WBS+
PS+
CM1
CM2
Influent
Effluent

7.0 Results: Plant Growth Trials

Carex secta grew equally well in all media, producing large plants within 6 months (Table 21, Fig. 35). Variability in basal diameter was higher for CM2, probably as a result of increased stress in some pots that were slow to fully rewet after drying, and hence greater drought stress. Similar wet: dry mass ratio indicates the proportion of dead leaves was similar in all treatments, so drought stress was not great enough to cause leaf death.

Table 21 Growth indices for *Carex secta* at harvest.

	Dry mass (g)	Basal mass (0 - 200 mm) (g)	Wet:dry mass (calc)	Total dry mass: basal dry mass	Basal Diameter (mm)
90% East Coast Sand +10% Compost B	35 ± 10	21 ± 5	2.8 ± 0.2	1.1 ± 0.3	35 ± 3
90% Pumice Sand + 10% Compost B	39 ± 6	20 ± 5	2.5 ± 0.2	1.2 ± 0.2	33 ± 4
CM1	29 ± 9	17 ± 7	2.7 ± 0.2	1.5 ± 0.5	32 ± 11

Austrofestuca littoralis grew similarly in both sand-based media (Table 22, Fig. 36). East Coast Sand washed from the base of some pots on first irrigation, requiring replanting of some replicates. In the main experiment, plants were infected with a root-sucking insect which was more abundant in the highly organic commercial mix, leading to slightly lower total biomass, but not height or basal diameter. A second trial was established in December 2011 with the remaining 8 root trainers, to again compare growth in the East coast sand and commercial mixes. All plants had some senescent leaves, and some had slightly more than others (Fig. 36), but overall no differences in mean biomass were evident between the treatments (Table 22). Similar wet:dry masses indicates the proportion of dead leaves was similar in all treatments.

Figure 35 Representative pots of *C. secta* at harvesting of bioretention growth trial; from left to right, CM1, PS+ and ECS+.

Table 22 Growth indices for Austrofestuca littoralis at harvest.

	Dry mass (g)	Basal mass (g)	Wet:dry mass (calc)	Total dry mass:basal dry mass	Basal Diameter (mm)	Height (mm)
90% East Coast Sand +10% Compost B	23 ± 4	16 ± 4	2.2 ± 0.3	2.2 ± 0.3	25 ± 3	49 ± 2
90% Pumice Sand + 10% Compost B	26 ± 4	18 ± 4	2.4 ± 0.2	2.4 ± 0.2	29 ± 1	52 ± 2
CM1 (30% organic)	16 ± 5	11 ± 4	2.1 ± 0.4	2.1 ± 0.7	23 ± 3	48 ± 3
90% East Coast Sand +10% Compost B #	11 ± 3	8 ± 3	2.5 ± 0.3	2.5 ± 0.3	19 ± 3	41 ± 2
CM1 (30% organic) #	9 ± 1	7 ± 1	2.0 ± 0.2	2.0 ± 0.2	18 ± 3	42 ± 3

planted in December 2011

Figure 36 Representative pots of *A. littoralis* at harvesting of bioretention growth trial; from left to right, CM1, ECS+ and PS+.

The water holding supply of PS+ mix was compared with commercial mixes. Pumice has higher dry bulk density under a standardised compaction than both commercial mixes tested, and a much lower volume of very large pores that are air-filled between rain events (27% vs. 37 to 40% for the two commercial mixes). However, the volume of stored water that plants can access for growth is similar in all substrates, being 21 to 24% of the total soil volume. This is because the two commercial mixes have a large amount of water that is held very tightly (to the organic matter) and therefore inaccessible to plants.

At an installed media depth of 600 mm, approximately 120 - 144 mm of water per bioretention cell unit surface area could be stored by the media tested, whereas at 1000 mm media depth, 200 - 240 mm per unit surface area could be stored (Table 23). Assuming plants could access moisture from the entire profile, this should be an adequate moisture supply to sustain plants during Auckland summertime conditions with evapotranspiration loosely approximated at 1 - 5 mm day^{-1} (Fassman et al. 2010; Fassman and Stokes 2011). Greater total water storage capacity would be achieved through increasing the media's compost proportion, which would also increase runoff retention capacity.

Table 23 Water supply characteristics for the bioretention media. Pressure is kPa tension under which pore volumes were measured.

	Bulk Density (Tm^{-3})	**Total pore volume (saturation)**	**Water @ FC (10 kPa) (%v/v)**	**Total Plant available water (10 - 1500 kPa)**	**Water @ 1500 kPa (%v/v)**
90% Pumice Sand + 10% Compost B	0.76	66	27	21	5
CM1	0.62	73	40	22	18
CM2	0.48	78	37	24	13
CM1 (field)[#]	0.81	65	40	22	18

[#] Measured using cores taken from a bioretention in Albany that used this product

Acceptable growth rates can be achieved with sand-based, low organic content mixes over 6 months when a low rate of slow-release fertiliser is added. Chemical tests indicate plants might require additional nitrogen and phosphorus in the medium term (12 to 24 months) if growth rates equivalent to those of amenity areas are wanted, or if high C:N ratio organic mulches are used to supress weed growth, as these will temporarily remove nitrogen from the soil. However this conclusion is not supported by the leaching column tests, which shows export of phosphorus.

The need for identification of triggers for fertiliser intervention and/or acceptable intervention options (e.g. use of organic mulches, or slow release inorganic) should be investigated through field trials.

Weeds will germinate and grow on the surface of these mixes, so either weeding, or mulch will be required to maintain high amenity value until bioretention develops a full plant cover, as is standard practice.

It is unlikely the three sand-based mixes developed will be any more drought prone than existing commercial mixes with high organic contents, as all store similar volumes of plant-available water per unit depth, and root growth is unimpeded (physically) in all mixes at standardised compaction level applied.

8.0 Conclusions and Recommendations

8.1 Research Summary

Landscape and aggregate suppliers in the Auckland region provide a limited range of sands and composts that could be suitable for bioretention filter media, according to criteria established from the international literature. Experience during this study demonstrates that having a dialogue with suppliers is useful in determining product information such as quality control, availability and consistency of supply, and cost.

Compaction testing shows water content is an important property to consider when installing filter media as different water contents will produce significantly different densities of media under the same compactive effort. A higher density for a material of particular mass is reflective of less pore space, and hence a lower hydraulic conductivity. In coarse media, smaller pores should also increase storage of plant-available water, which also enhances runoff retention. In practice a balance between infiltration rate and water retention is needed. Compacted densities for media with a high proportion of compost can change by up to 26% as the water content is altered with infiltration rates being reduced by a factor of four. East Coast Sand and Woodhill Black Sand without organic content display no change in resulting compaction when water content is varied, indicating that the organic proportion is of most importance when it comes to compaction of sand:compost mixes. Most bioretention filter media will contain organic matter, and therefore it is important to specify a comprehensive compaction strategy which takes into account water content.

A hydraulic conductivity range of 12.5 to 150 mm hr^{-1} was established to meet objectives for limited ponding times and runoff to media contact times for pollutant removal. Based on comparison of candidate sand's particle size distributions against international bioretention design guidelines, No.3 sand from ALS (all passing 2 mm, U=3.0) was initially chosen to be the sand base for hydraulic tests. Hydraulic testing with homogeneous mixes based on No.3 sand found the sand to be too permeable (over 2,000 mm hr^{-1}) and unable to meet the target conductivity range. This result indicates particle size distribution guidelines should not be used as a substitute for hydraulic testing. Testing also shows the hydraulic conductivity of sand-based mixes does not significantly differ between light tamping compaction or wetting and settling compaction.

Testing media with finer sands portion (East Coast Sand [all passing 0.425 mm, U=1.4], Woodhill Black Sand [all passing 0.425 mm, U=1.8], Pumice Sand [all passing 2 mm, U=11.1]) mixed with 10% bark-based compost resulted in media requiring only wetting and settling compaction came relatively close to the target hydraulic conductivity. The mean hydraulic conductivities for the three finer sand based media were 80, 400, and 340 mm hr^{-1}, respectively. Both commercial rain garden media (CM2 and CM1) were close to the target hydraulic conductivity range when mechanical light tamping compaction was applied being 240 and 180 mm hr^{-1}, respectively.

From hydraulic testing results, five mixes and compaction treatments were chosen for water quality testing:

- East Coast Sand + 10% Compost B [wetting]
- Woodhill Black Sand + 10% Compost B [wetting]
- Pumice Sand + 10% Compost B [wetting]
- Two commercially available, proprietary bioretention mixes [lightly tamped]

Water quality testing involved dosing media with synthetic stormwater containing realistic concentrations of dissolved zinc, dissolved copper and soluble phosphorus. Media were also aged to 5 years, 10 years, and 15 years using concentrated doses. Quantitative results of media aging should be interpreted for comparative purposes, rather than as predictive indications of field concentrations or removal efficiencies.

There was no correlation found between runoff to media contact time and pollutant removal efficiency. Dissolved copper and zinc were removed by the media, therefore the result indicates the range of tested hydraulic conductivities (80 to 400 mm hr^{-1}) is low enough for adsorption (both specific and non-specific) to occur. Similarly, no effect was found when varying drying times between doses by up to five days.

Rough estimation of cumulative pollutant mass loading over 15 simulated ‘years’ suggested sand based media could remove around 60% of dissolved copper loading and 70% of dissolved zinc loading. CM2 is estimated to remove 36% and 46% of dissolved copper and dissolved zinc loading respectively.

CM1 is estimated to remove 53% dissolved zinc loading, but leach 15% extra copper over 15 years. The reason for dissolved copper leaching could be due to the media already containing substantial amounts of copper, or because of the formation of dissolved organic carbon in the organic portion, which in turn facilitates copper mobility in soils (Altaher 2001).

All media trialled except CM1 consistently reduced typical stormwater dissolved copper concentrations below 5 μg/L, and zinc concentrations below 10 μg/L. The removal was replicated for media of 0, 5, 10, and 15 ‘years’ of aging, with no signs of deteriorating performance. CM1 initially displayed leaching for copper and only slight removal for zinc, possibly due to an initial saturation of heavy metals and/or leaching facilitated by complexes of copper with dissolved organic carbon. As testing progressed, CM1 showed improved effectiveness at reducing heavy metal concentrations.

For phosphorus, all media trialled exhibited substantial leaching within 5 'years' of media aging. Effluent phosphorus concentrations were often two or three orders of magnitude higher than influent. CM2 effluent contained the greatest concentrations of phosphorus (3,200 μg/L for media aged 5 and 10 years, 2,300 μg/L for 15 years), while CM1 contained the lowest (550 μg/L). Media chemistry analysis suggests that fertilizer phosphorus is added to both commercial rain garden mixes. Effluent from East Coast Sand based media had phosphorus concentrations of 600 μg/L at 5 years of aging, increasing to 1,300 μg/L at 10 years. Effluent from Woodhill Black Sand based media similarly increased from 1,000 μg/L to 1,400 μg/L after 10 years. Effluent from Pumice Sand based media was consistently leaching 1,200 μg/L across 15 years. The effluent concentrations are extremely high when compared to influent concentrations of65 μg/L.

Media chemistry analysis indicated that the compost component in the 90% sand: 10% compost mixes is the vital component for providing the CEC required for heavy metal removal in the absence of iron or aluminium based materials (e.g. 'baseball field clay') or other component with high-surface area that does not compromise infiltration rate or resilience to compaction. The importance of sand in these cases is to physically stabilize the system and provide an adequate water retention time for the compost components to perform chemical pollutant removal, as well as providing physical filtration to remove contaminants attached to the filtered sediment.

Biomass accumulation and vigour of two bioretention plants, *Carex secta* (wet tolerant) and *Austrofestuca littoralis* (drought tolerant) were measured after 6 months of growth in three bioretention mixes: 90% East Coast Sand + 10% v/v bark-based compost, 90% Pumice Sand + 10% bark-based compost and a CM1 with c. 30% compost. Plant species grew satisfactorily in all bioretention mixes. The East Coast Sand mix exhibited some crusting in the short term, whereas the Pumice Sand mix and commercial mix did not show crusting. Grasses and herbs germinated on all bioretention mixes.

The pumice sand and two commercial mixes stored similar volumes of plant-available water (measured at 10 - 1500 kPa tension) Water holding capacity should be sufficient to sustain plants subject to Auckland's summer evapotranspiration, assuming plants can access moisture over the full soil profile.

8.2 Consideration of International Guidelines and Recommendations

Limiting a specification to PSD is considered insufficient to ensure appropriate hydraulic conductivity. A coarse sand (all passing 2 mm, with $U \approx 3$) or high silt/clay component (considered to be >20%, as might be found in natural soils) will be susceptible to the field installation compaction method, and hence so will be the hydraulic conductivity. In practice, if these materials are used, careful installation procedures and post-installation testing of infiltration and/or bulk density would be strongly recommended.

None of the media mixes trialled herein completely satisfies recent international design recommendations for bioretention media. It is noted that the methodology for measuring hydraulic conductivity used herein varies from the ASTM methods by FAWB, WSU, and SPU (see Section 4.5) which may have some influence on results. Nonetheless, mixes that satisfied aggregate PSDs tended to produce extremely high hydraulic conductivities, even when mixed with relatively high levels of compost (which also violates many international guidelines). To achieve assumed appropriate hydraulic conductivities, aggregate gradation and/or uniformity was compromised, potentially posing a risk for structural failure. The moisture-density curve for a sand mix that meets international PSD guidelines indicates low risk of structural collapse under loading; these tests should be repeated for the 90% sand: 10% compost mixes that did not meet the PSD range, but more successfully achieved the target hydraulic conductivity. A mix that is resilient to compaction will likely have a fairly consistent hydraulic conductivity.

The intent of limiting hydraulic conductivity is to achieve removal of pollutants that require adequate contact time for attenuation and to achieve adequate water storage potential to sustain plants. Despite somewhat high hydraulic conductivity of the 90% sand: 10% compost mixes, the limited scope of laboratory testing suggested performance for heavy metals' removal was not time-dependent (i.e. it was independent of hydraulic conductivity). Plant pot-trials and limited media moisture release data indicated adequate moisture storage capacity; however, field testing would be valuable.

Organic matter used in bioretention applications must balance nutrients for plant growth against leaching potential. Rapid, lush plant growth is generally seen as desirable (high amenity) and is achieved in the landscaping industry by providing abundant levels of the macro-nutrients nitrogen and phosphorus, both of which are also potential surface water pollutants that bioretention can be designed to attenuate.

Identifying an appropriate media mix for bioretention cells is inherently limited to available materials with high quality assurance. At the same time, there is scope for industry development of materials that are fit-for-purpose. Of particular concern for bioretention application is compost composition and the propensity for phosphorus and copper leaching. The current New Zealand standard for maximum copper concentrations in compost (Standards New Zealand 2005) is likely too high to prevent a 'first flush' where composts are used as a high proportion of bioretention mixes. The majority of Auckland's receiving environments are not likely phosphorus-sensitive as they are short, steep catchments leading to estuarine and/or marine outlets. Nonetheless, where freshwater environments do exist, for example, Lake Pupuke, neither the composts nor commercially available bioretention mixes evaluated herein would be suitable. Several New Zealand-sourced materials for enhancing pollutant removal were noted in the literature review (Section 2.1 and 3.2), but there is no publically available research in stormwater treatment applications. There is a growing body of evidence for using drinking water treatment residuals (a waste product) for phosphorus removal (O'Neill and Davis 2012 a, b). Any additives or amendments need to maintain resilience of the rain garden media to physical compaction to ensure adequate permeability, and the amendment needs to have consistent properties. It is noted that logistics precluded nitrogen testing in testing in the laboratory; this nutrient is of concern in the Hauraki Gulf (Hauraki Gulf Forum 2011).

Laboratory predictions do not necessarily yield similar field results. While some additional laboratory investigation may be worthwhile, field trials of sand-based, low organic-matter bioretention mixes in combination with organic mulches are strongly recommended. At a minimum, a field investigation of a non-proprietary media should be scoped to include heavy metals and nutrient analysis (nitrogen and phosphorus), as well as water flow and storage characteristics (hydrology).

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