

Reconstruction of coal tar-contaminated roads by in-situ recycling using foamed bitumen stabilisation

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Abbreviations and acronyms

AADT	Annual average daily traffic
CBR	California-bearing ratio
DOC	Dissolved organic carbon
FB	Foamed bitumen
FID	Flame ionisation detection
IANZ	International Accreditation New Zealand
ITS	Indirect tensile strength
MI	Mutagenicity index
PAH	Polycyclic aromatic hydrocarbon
TLA	Territorial local authority
TPH	Total petroleum hydrocarbons
US EPA	United States Environmental Protection Agency

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Executive summary

Coal tar binders were widely used for road construction in New Zealand until the 1970s. Consequently a large proportion of older city streets still contain carcinogenic coal tar in subsurface seal layers. For example in Christchurch, it is estimated that ca 50% of the city's streets are affected, and that in the next 10–20 years many of these streets will require reconstruction. Previous research has shown seal layers contain ca 4000mg/kg of polycyclic aromatic hydrocarbons (PAHs), and that tar-contaminated aggregate is toxic to the soil-dwelling insect *Folsomia candida*, with a reported LC₅₀ (lethal concentration to 50% of test population) value of ca 300mg/kg. With such high concentrations of PAHs and potential for toxicity, conventional reconstruction methods would require costly landfill disposal of any excavated tar-contaminated material. Such a requirement, however, would most likely be prohibitively expensive, and therefore research was conducted to evaluate the merits of in-situ recycling of the tar-contaminated material as an alternative to excavation and landfill disposal. The in-situ process involves stabilising the pulverised material with foamed bitumen (FB) (and usually cement) and the compacted mixture forming the construction base on the new road. The hypothesis was that adding FB to pulverised material would reduce the leaching of PAHs, resulting in a relatively low-risk material with little potential to harm the environment. The research was undertaken in two stages:

Stage 1 – identification of suitable street, optimum design mix established by testing physical properties (eg indirect tensile strength) and contaminant mobility in stabilised samples (using laboratory prepared samples)

Stage 2 – field stabilisation trial on Burke St; and contaminant leaching and toxicity evaluations of control and different stabilised materials.

Stage 1: Preliminary investigation

The four selected Christchurch streets (ie Newcastle, Ryan, Burke and Glasgow) were sampled at four locations to confirm the presence of coal tar (via PAH concentration). The mean PAH concentrations in the seal layer of Newcastle, Ryan, Burke and Glasgow were 870, 1230, 2380 and 910mg/kg, respectively, confirming that all four streets contained coal tar binders in subsurface seal layers. Although not containing the thickest layer of total construction material (seal + basecourse), Burke St (initially estimated with a thickness of 105mm) was selected as the trial site because it was considered more representative of typical coal tar-contaminated streets.

The 250m section of Burke St selected for the trial was further characterised by Benkelman Beam testing and California bearing ratio (CBR) measurements performed at eight bore holes located along the test section. The CBR is simply the ratio (expressed as a percentage) of the pressure required to penetrate a soil sample with a plunger of standard area divided by the pressure required to achieve an equal penetration on a standard crushed rock material. Hence a CBR of 5% (typical for moist clay; moist sand is ca 10%) means the required pressure to penetrate the tested sample is 20-times less than it is for the standard crushed rock material. The CBR values for the subgrade material were >15% at six of the eight locations, but were less <5% at two bore holes at the most western end of the trial section. Bore hole investigations resulted in the initial thickness estimate of 105mm being revised to 130mm, which, based on assumed traffic loads, was sufficiently thick to produce a rehabilitated pavement with a >25-year service life (where subgrade CBR >15%).

A series of design mix formulations of Burke St material containing 1%, 2%, 3% and 4% FB (all containing 1.5% cement) were prepared and tested for indirect tensile strength (ITS) and contaminant leaching. Relative to unstabilised material, samples containing 4% FB reduced the leachate concentrations of major PAHs (ie pyrene, fluoranthene and acenaphthene) by a factor of 2–3. Stabilisation using 1% or 2% FB was less effective, and actually resulted in increased PAH leaching (relative to the control) because of the alkaline conditions caused by the cement (1.5% loading). Increased alkalinity was strongly correlated to the concentration of dissolved organic carbon (DOC), which in turn was responsible for solubilising more PAHs as well as non-tar contaminants such as copper. For higher FB loadings (4%), the increased proportion of bitumen appeared to override the enhanced PAH solubilisation effect of the leachate DOC.

As a result of stage 1 investigations, the decision was made to proceed with the full stabilisation of Burke St. The recommended rehabilitated pavement plan consisted of a 130mm FB (3%) + cement (1.5%) stabilised base with an overlain 30mm layer of chip seal.

Stage 2: Field trial

The field trial was carried out on 18 March 2008, using a Wirtgen 2500 in-situ recycling machine. Experimental samples for leachate testing comprised i) unstabilised, ii) 'cement only' stabilised, and iii) 'FB + cement' stabilised, which were collected from two 10m long test sections near the eastern end of Burke St (near the Antigua St intersection). The stabilised materials typically contained 750–1000mg/kg of PAHs.

Leachate pH for samples containing cement was ca 12, and this was strongly correlated to DOC concentration ($R^2 = 0.97$). Leachate testing was conducted using 1kg of material (compacted and non-compacted) and 2L of water, corresponding to a liquid-to-solids ratio (L/S) of 2. The concentration of PAHs solubilised under the leaching conditions ranged from 10 to 52 $\mu\text{g/L}$, with the highest concentration from non-compacted (or loose) 'cement only' material. The inclusion of 3% FB reduced the amount of PAHs solubilised four- to five-fold (to 10 $\mu\text{g/L}$ of PAHs), but compaction of FB-stabilised samples had very little effect on PAH immobilisation (decrease from 10 to 6–7 $\mu\text{g/L}$). Compaction of cement-only samples resulted in a two- to three-fold decrease (from 52 to 19 $\mu\text{g/L}$) because less cement was available for leaching, resulting in less alkaline conditions (10 cf to 12) and therefore less DOC for solubilising PAHs. In general, the leachate concentrations of PAHs and the influence of additional bitumen in stabilised samples were consistent with equilibrium partitioning of PAHs between aqueous and bitumen phases.

The toxicity of leachates from Burke St road materials was determined via a growth inhibition assay using the fresh water algae, *P. subcapitata*. Despite the concentrated nature of the extractions, the potential for harm of the leachates varied from 'low' to 'moderate' – in both cases the moderate toxicity corresponded to loose samples that contained cement. The inclusion of FB had no effect on toxicity, indicating that tar-derived PAHs were not the cause of the observed toxicity – this was consistent with the highest individual concentration (6 $\mu\text{g/L}$ of fluoranthene) for Burke St leachates being 30 times below literature toxicity (EC_{50}) values. It was interesting to note that leachates from unstabilised pulverised Burke St material had a low potential for harm, which suggests there would be minimal risk of leachate toxicity if this type of material was removed and stockpiled offsite. However, the attraction of stabilisation is that allows the material to be compacted into a semi-bound form and reused as road base material – effectively locking it away from any losses/exposure to any potentially sensitive environmental compartments.

Leachate toxicity was strongly correlated ($R^2 = 0.94$) to original pH of the solution (note: pH was adjusted to ca 7 immediately prior to algal testing) indicating that toxicity was attributable to heavy metals. The total concentration of seven heavy metals was 310–350 $\mu\text{g/L}$ in cement-containing samples, and 85–105 $\mu\text{g/L}$ in compacted samples and unstabilised controls. Chromium and copper accounted for 90% of the heavy metals total, but the observed toxicity was most consistent with leachate copper being largely

responsible. The experimentally determined copper EC₅₀ values for Burke St samples exhibiting moderate toxicity were 14–18µg/L, which were in agreement with literature concentrations of 15–25µg/L. Therefore assuming the source of heavy metals is aggregate/subgrade material, the observed leachate toxicity is not associated with the presence of coal tar. That is, in the absence of any tar contamination, recycled materials stabilised with cement would be expected to exhibit similar toxicity. Based on these findings, the reuse of tar-contaminated pavements as stabilised road base material represents no more risk to the environment than recycling conventional bitumen-only pavements.

The major limitation of applying the in-situ recycling technique to tar-contaminated roads in New Zealand is that the affected streets are typically composed of thin construction layers. Unless it is possible to apply sufficient make-up aggregate, the stabilised road base will be of substandard strength and prone to early failure. Despite preliminary investigations, the Burke St stabilisation carried out with a recycling depth of 130mm produced a stabilised road base with a very silty/dusty surface that prevented good adhesion between the base and chip seal wearing course. The result: nine months later Burke St had a number of patches where the chip seal had delaminated, exposing the tar-containing road base, and increasing the chance for losses of this material to the environment (roadside soil, aquatic receiving environments, etc).

Despite this limitation, the research has shown that although the tar-contaminated material contained high concentrations of PAHs, the hydrophobic nature of these contaminants meant there was little risk from leaching. Hence, it is very much an environmentally responsible option to reuse this material in ecological 'irrelevant' areas such as beneath carriageways. The alternative of transporting and landfill disposal is not only economically unfeasible but, depending on the distance to nearest landfill, can also mean a high environmental cost (energy, CO₂, etc).

Abstract

Coal tar-derived roading material contains over 1000 times more polycyclic aromatic hydrocarbons (PAHs) than equivalent bitumen pavements and has been identified as a major source of PAHs in both Christchurch and Auckland aquatic receiving environments. Many old streets containing coal tar will soon require reconstruction, and therefore the excavation and potential disposal of contaminated road construction layers represents a significant financial and environmental problem. To address this problem, we evaluated in-situ foamed bitumen (FB)/cement stabilisation as an environmental acceptable method to reuse the contaminated tar road material.

Based on contaminant leaching and toxicity, the reuse of tar-contaminated roads as compacted stabilised base material represents minimal risk to the environment. FB decreased PAH leachate concentrations by ca 4–6, although algal toxicity was correlated to leachate copper, which was increased by the co-use of alkaline hydraulic binders. Despite this, the low potential for harm from the leachates combined with a reuse application (ie road base) that limits environmental exposure/risk via being: i) capped with a water-proof seal layer; and ii) located beneath carriageway and thus not being reused in an environmentally significant 'compartment'. A limitation of the method is that many of the older tar-contaminated streets may not be suitable for in-situ FB recycling without additional makeup aggregate being applied.

1 Introduction

The aim of the research was two-fold; first to analyse the extent of the issues relating to tar-contaminated roads, and second, to trial and test in situ foamed bitumen/cement stabilisation techniques to immobilise potentially toxic tar-derived PAHs.

1.1 The coal tar problem and the need for the research

Coal tar-based binders are a by-product of coal gasification (ie gas works), a process that involved heating coal in large retorts at temperatures of 800–1200°C. During the process, crude coal tar was distilled from the retorts with carbonised coke remaining. The crude coal tar was then distilled into numerous fractions (or products) with the tar residue being used to make various grades of road tar binders (and hence the term ‘tar sealing’). Relative to bitumen-based binders derived from crude oil, coal tar contains ca 10,000-fold higher concentrations of polycyclic aromatic hydrocarbons (PAHs). Numerous studies have established that PAHs have carcinogenic, mutagenic and teratogenic effects on animals (Hoffman and Wynder 1971; Grimmer 1983; Perera 1997). This is reflected in the mutagenicity index (MI) (via Modified Ames test), of the two binder types. Blackburn et al (1996) have reported MIs of uncracked petroleum oils (bitumen precursor) of around 20, while in contrast typical values for coal tars range from 300 to well over 1000. In addition to human health effects from exposure to fumes from hot tar-containing road binders, tar-derived PAHs released into the environment can accumulate in aquatic sediments, where they can pose a long-term environmental risk to aquatic organisms (Beasley and Kneale 2002). Previous research by the National Institute of Water & Atmospheric Research (NIWA) has shown that coal tar is the major source (up to 95%) of sediment-associated PAHs in numerous urban aquatic receiving environments in Auckland and Christchurch (Reed and Depree 2008; Depree and Ahrens 2008).

Previous research has confirmed the presence of tar-contaminated roads in Auckland and Christchurch (Depree and Olsen 2005; Depree 2006; Ahrens and Depree 2006), but the reality is that any street constructed prior to ca 1970 that has not been reconstructed probably contains coal tar in subsurface layers. This corresponds to approximately half the streets in Christchurch City, and no doubt affects a large number of other cities and towns throughout the country; however, the general awareness in New Zealand of the presence and associated issues (environmental, human health and safety) of coal tar in roads is very low. Presumably this is attributable to the tar being buried below bitumen layers and is therefore a case of ‘out of sight, out of mind’. But with the pending reconstruction of many affected roads, coal tar issues are starting to resurface – literally – with tar-contaminated subsurface layers being unearthed and hence leading to the potential for exposure to humans and the environment.

In Europe, the ‘lid is already off Pandora’s box’ regarding the issues of tar-contaminated asphalt, with many countries having introduced regulations to limit recycling of tar-contaminated road material into hot mix asphalt. In these countries, excavated or milled material exceeding the maximum limit must be disposed of in a suitable landfill. Although there is currently no definition of tar-contaminated road seal in New Zealand, reclaimed asphalt containing more than 0.1% coal tar is considered a hazardous waste in Europe (EAPA 2005). Furthermore, in another working document on landfill waste, it has been proposed that bituminous mixtures must contain <25mg/kg to be considered inert waste (EAPA 2005). With reported PAH concentrations of 1500–4000mg/kg in New Zealand road seal layers, (Depree and Olsen 2005), there is clearly a need for some national guidance on how to best manage tar-contaminated road materials.

If New Zealand adopted definitions similar to those in Europe regarding 'hazardous bituminous mixtures' (eg >25mg/kg considered hazardous waste), it would require disposal of tens of thousands of tonnes of excavated tar-contaminated material as hazardous waste, adding millions of dollars to road reconstruction projects. In order to avoid such a prohibitively expensive directive in the future, the New Zealand transport sector needs to be proactive and develop economically feasible and environmentally responsible alternatives for managing the issues of reconstructing tar-contaminated roads. With a national waste strategy that clearly prioritises recycling and reuse options over landfill disposal, there is considerable scope to utilise existing road recycling techniques to manage the future waste stream of tar-contaminated road materials in New Zealand. Such techniques will enable the tar-contaminated material to be safely recycled into a compacted road base layer overlain with a bituminous wearing course – effectively isolating the coal tar contaminants without the need for the high cost (financial and environmental) of landfill disposal. Keeping the coal tar in its current location is beneficial because the contaminated material will be sealed beneath bitumen layer, which will prevent any leaching or erosion of contaminated material. Second, the area in which the contaminated material is being used (construction material beneath carriageway) is not an environmentally relevant compartment (with respect to living organisms) and as such, responsible reuse of tar-contaminated material represents minimal risk (considering the coal tar has been present for >30–40 years).

Based on recycling/stabilising methods commonly used in New Zealand (refer to section 1.2.7), and the aims to i) minimise transportation, exposure and handling of tar-contaminated material, ii) effectively bind the pulverised tar-contaminated material in a compacted, water impervious and flexible layer (to minimise cracking), and iii) reuse material in existing carriageways as road base material for new roads, in-situ recycling using foamed bitumen (FB) stabilisation was a potentially suitable method for re-constructing tar-contaminated roads.

The aim of the research was to evaluate in-situ foamed bitumen recycling as an economically feasible, practical and environmentally responsible solution for managing coal tar-contaminated road materials in New Zealand. To this end, a combination of laboratory and field stabilisation tests was undertaken using tar-contaminated road materials from a Christchurch street (Burke St). Stabilised materials were subjected to batch leaching tests in accordance with established Norwegian protocols, and leachates were analysed for contaminants and toxicity.

The project was carried out between July 2007 and June 2008 and was composed of seven components spread over two research stages. Stage 1 largely involved identifying a street suitable for field-scale stabilisation and preliminary contaminant leaching experiments. Stage 2 involved field stabilisation trials in Christchurch using a Wirtgen 2500 recycling machine and testing the materials for contaminant mobility and leachate toxicity to determine the overall merits of in-situ FB stabilisation for managing tar-contaminated road materials in New Zealand.

1.2 Coal tar: additional information and literature review

1.2.1 Coal tar use: New Zealand

Coal tar binders were used extensively in road construction throughout New Zealand prior to the 1970s, with National Road Board technical specifications existing for a range of tar products, from low viscosity tar primers through to high viscosity roading tars for chip-sealing applications (NRB 1974). Coal tar was often used as primer, a high penetration grade (low-viscosity) binder applied to the surface of the base

course to ensure good adhesion of subsequent chip seal layers, which were either bitumen- or coal tar-based layers. Because coal tar has not been widely used for at least 30–40 years in New Zealand, the coal tar seal layers have been covered by one or more bitumen chip seal or asphalt layers. Previous research by NIWA has confirmed the presence of very high concentrations of PAHs in subsurface seal layers in roads and footpaths taken from older residential areas of Auckland and Christchurch (Depree and Olsen 2005; Ahrens and Depree 2006). To give an idea of the potential size of the ‘tar problem’, Christchurch City Council estimates that up to 50% of the city’s streets may still contain coal tar binders.

1.2.2 Coal tar use: European examples

Many European countries and the United Kingdom have a history of using large quantities of coal tar in road construction, and as in New Zealand, a large proportion of roads still contain coal tar in either subsurface or surface layers. Adding to the problem in Europe is that, in addition to tar binder in very old (ie deeper) layers of roads, coal tar was used in the production of asphalt paving until the early 1990s. Such countries are now faced with the challenge of managing roading networks that contain millions of tonnes of tar-contaminated asphalt. For example, in the Netherlands, between 1963 and 1991, on average 10,000 tonnes of coal tar binders were used annually in road construction, and ca 25% of the roading system still contains coal tar in the wearing course (Bulk and van der Zwan 2000). In Sweden, coal tar-containing asphalt was used from the late 19th century until ca 1973, and approximately 20 million tonnes is present in public roads (Andersson-Sköld et al 2007). In Switzerland, coal tar binders were used extensively in asphalt pavements up to ca 1990, during which time the commonly used asphalt binder contained 5–20% coal tar (Hugener et al 2004).

1.2.3 Coal tar use: United States of America (sealcoat)

In addition to being used historically in road construction, the major issue in the United States is the continued practice of applying coal tar-based sealcoat to protect asphalt surfaces (eg parking lots) and also to maintain the aesthetic black appearance (eg homeowners’ driveways). To maintain protection/aesthetic properties of the asphalt surface, sealcoats are typically reapplied every one to three years. Tar-based sealcoats (containing 15–35% coal tar) are often preferred because of their superior properties (ie fuel resistance), and as such are used in all 50 states, although their use is more prevalent east of the North American continental divide (Van Metre et al 2009).

In the few states where data is available, the amount of coal tar seal coat being used is extremely high. For example, in the State of Texas it is estimated that 225 million litres of tar-based seal coat are applied annually (Scoggins et al 2007). In the New York harbour watershed, annual application rates of tar-based sealcoats have been estimated at 5.3 million litres (Valle et al 2007). Mahler et al (2005) reported that coal tar sealcoat use was the main source of PAHs to stream sediments in four watersheds in Texas. As a result, since January 2006, there has been a complete ban on the sale and application of tar-based sealcoats in the city of Austin. Coal-tar sealcoat sales and use are also now banned in Dane County, Wisconsin and the city of Washington DC. More recently, Van Metre et al (2009) have indicated the potential size of the coal tar sealcoat problem in the United States, observing a link between catchment coal tar sealcoat use and elevated PAH concentrations in lake sediments from urbanised catchments in nine cities across the United States.

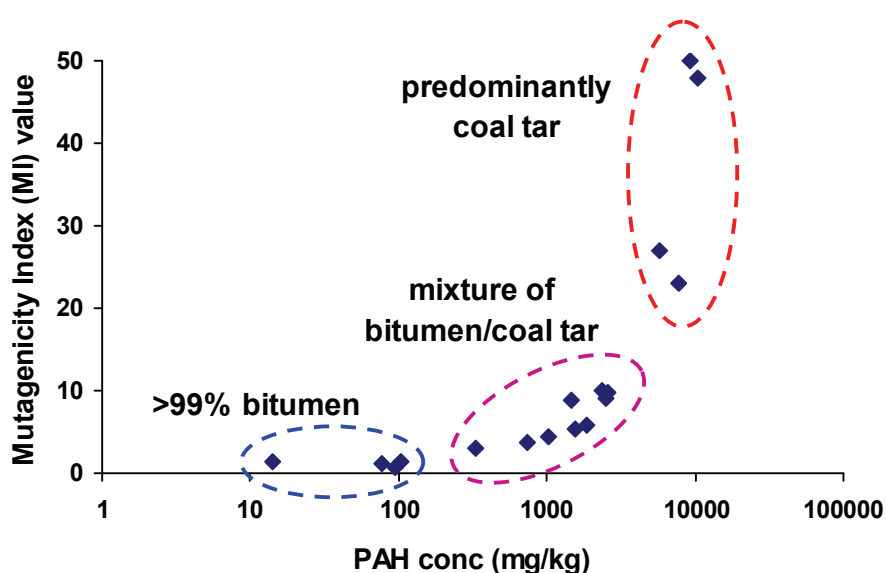
1.2.4 Toxicity of tar-contaminated roading materials

The European waste catalogue and hazardous waste list indicates that reclaimed asphalt containing more than 0.1% coal tar should be regarded as a hazardous waste (EAPA 2005). Assuming a typical coal tar binder content of 150,000mg/kg PAHs, the 0.1% hazardous definition corresponds to ca 150mg/kg of PAHs.

The Asphalt Institute undertook a study of road cores in Copenhagen to determine the toxicity of road materials and whether the observed toxicity (measured via the modified Ames test; ASTM method E1687) was attributable to the presence of coal tar-derived PAHs (Blackburn et al 1999). A series of cores were taken, separated into different seal layers, and analysed for PAHs (16 US EPA) and the mutagenicity index (MI) determined via the modified Ames test. In general, materials with MIs of 1 or less are not carcinogenic, and those with MIs greater than 1 are considered carcinogenic. Carcinogenic potency increases linearly with MI values up to about 50; at which point maximum potency in the skin painting assay is observed (Petrolabs 1998).

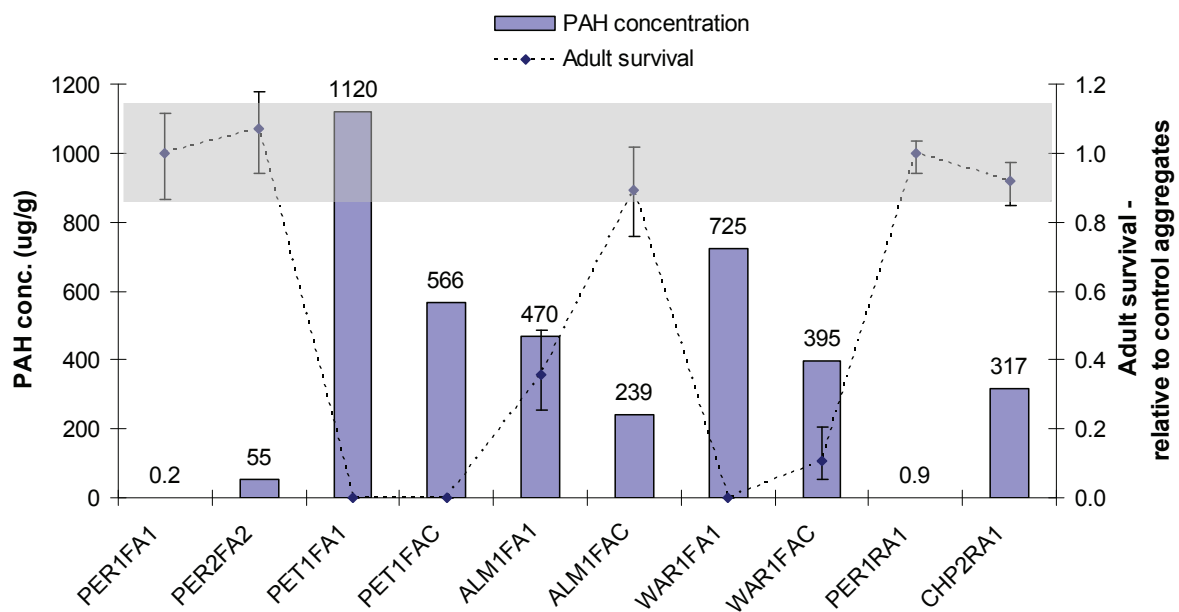
Selected results from the Blackburn et al (1999) study are summarised in figure 1.1. The key points are that samples containing predominately bitumen (<100 mg/kg of PAHs) had very low MI values (ca 1), and predominantly coal tar materials (ca 10,000 mg/kg of PAHs) had highly carcinogenic MI values of up to 50 (which corresponds to maximum potency in the skin assay). Although not obvious from the logarithmic concentration axis in figure 1.1, the mutagenicity indices were strongly correlated ($R^2 = 0.95$) with the concentration of PAHs in the road materials (Blackburn et al 1999). Although a value of >1 is indicative of a potential carcinogenic material, the data was more variable at lower PAH concentrations. As a guide, and assuming similar PAH compositions across coal tar binders, the concentration of PAHs in road material that corresponded to MI values of 1, 2 and 3 were 140mg/kg, 370mg/kg and 600mg/kg, respectively.

Figure 1.1 Mutagenicity indices versus concentration of PAHs from Danish mastic asphalt road cores. Logarithmic scale for PAH concentration was used to give more separation of low PAH concentration data points. Figure plotted using selected data published by Blackburn et al (1999)



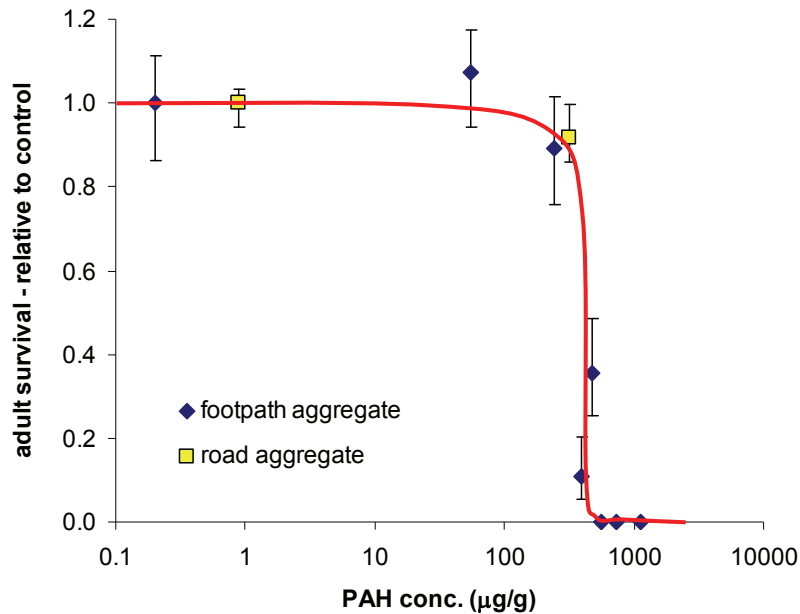
NIWA and Christchurch City Council (Deprea 2007; Deprea et al 2008) have carried out toxicity studies on tar-contaminated aggregate using the soil-dwelling insect, *Folsomia candida* (figure 1.2). Tar-contaminated aggregate containing >560mg/kg resulted in 100% mortality, samples below ca 300mg/kg of PAHs showed no observed toxicity, and samples containing ca 400mg/kg of PAHs resulted in partial mortality of *F. candida*.

Figure 1.2 PAH concentration and acute toxicity to *F. candida* of tar-contaminated aggregates from Christchurch roads (from Deprea 2007) – samples PER1FA1, PER2FA2 and PER1RA1 were controls



The sharp transition between non-toxic and toxic response of *F. candida* to PAH concentrations in tar-contaminated aggregates is more apparent in the dose-response plot (figure 1.3). The transition occurs over a window of only ca 100mg/kg of PAHs, with samples containing <300mg/kg of PAHs being non-toxic, and samples containing >400mg/kg resulting in almost 100% mortality. This corresponded to an LC₅₀ (concentration required for 50% mortality, relative to controls) of 301mg/kg (95% confidence interval of 270–340mg/kg), which was consistent with the 140–600mg/kg concentration range of samples exhibiting MI values of 1–3 (Blackburn et al 1999).

Figure 1.3 Dose-response plot of adult survival of *F.candida* vs concentration of PAHs in tar-contaminated aggregate (Depree 2007)



1.2.5 Alternatives to excavation and disposal of coal tar

In accordance with the New Zealand Waste Strategy (MfE 2004), the waste hierarchy specifies the most preferred option is reduce, followed by reuse, recycling, recovery, treatment and, lastly, disposal. As such, reuse and recycling options for tar-contaminated materials should be considered prior to the material being disposed of in a landfill. This is particularly important, because when total environmental costs (ie transport emissions from exporting old materials and importing new materials) are taken into account, landfill disposal of tar-contaminated material may not be the most environmentally effective course of action. A Swedish study (Andersson-Sköld et al 2007) entitled *Coal tar-containing asphalt – resource of hazardous waste* analysed the cost (economic and environmental) of different scenarios for managing tar-contaminated material, which included reuse on-site (as stabilised road base), bioremediation, combustion and landfill. For one of three sites evaluated, the cost for disposal was €3 million compared with onsite stabilisation costs of only €10,000. At the other two sites, the absolute costs for landfill disposal were lower (ie less material and closer proximity to landfill), but were 50- and 100-fold more expensive than if reusing the material onsite. Furthermore, the energy use (measure in litres of diesel per m² of road) was 0.13L diesel/m² for on-site emulsion stabilisation, compared with 3.19L diesel/m² for landfilling the tar-contaminated material – higher by a factor of 24. Based on the results of the desktop study, the authors concluded that reusing tar-containing material in new road construction was the most favourable alternative in terms of cost, material use, land use, energy consumption and air emissions.

Given the toxicity of tar-contaminated materials (section 1.2.4), options for reusing or recycling this material should: i) reuse the material in the carriageway; ii) minimise exposure time of material to the environment and human workers during processing; iii) encapsulate or bind the tar-contaminated material into a compactable and water impervious matrix in order to mitigate leaching of contaminants and

migration of tar-containing particulate material; iv) be cost effective and based on existing methods, equipment and capability; v) address both tar contamination in the seal layer and the underlying aggregate (crushed stone) commonly referred to as basecourse; and vi) minimise transportation of materials offsite and maximise onsite or in-situ techniques. Potentially useful recycling and reuse applications of tar-contaminated materials are discussed in the next sections.

1.2.5.1 Recycling tar-contaminated road material into asphalt

General

Asphalt is one of the most recycled materials in the world, with some countries like the United States claiming recycling rates of up to 80% (Smith 2003). Large amounts of coal tar are still present in many European roads, and much is incorporated in road millings (or planings) when reconstruction work is carried out. Normally, the road millings would be used to make new asphalt that typically contained 20–40% recycled asphaltic pavement (Hugener et al 2004). The two main concerns about recycling tar-contaminated asphalt are the safety of workers exposed to tar fumes (particularly hotmix applications) and the contamination of increasing quantities of asphalt. These concerns have been addressed in different ways, ranging from uncontrolled use in southern and eastern countries to effectively a complete ban in the Netherlands (Hugener et al 2004).

Recycling of tar-contaminated asphalt in Europe

In the Netherlands it is estimated that of the three million tonnes of asphalt recovered annually, up to 500,000 tonnes contains coal tar binders. To protect the quality of the soil and water (surface and ground), the Building Materials Act (since 2000) specifies a maximum limit of 75mg/kg (sum of 10PAH) for any stone material, including recycled asphalt, that is used on or in the soil. With PAH concentrations of up to 10,000mg/kg, this regulation effectively prohibits recycling tar-contaminated asphalt, and as such, these materials are now incinerated using a special process (Torbed® technology) that burns off the tar/bitumen binder leaving behind the aggregate for recycling (Bolk and van der Swan 2000).

Switzerland has fewer strict laws and regulations for recycling tar-contaminated asphalt, allowing hot-mix recycling of tar asphalts containing up to 20,000mg/kg of PAHs in the binder (sum of the 16 US EPA), which equates to an asphalt PAH content of ca 1000mg/kg. Because of high disposal costs and limited budgets for road maintenance, road authorities in Switzerland would like to be able to recycle tar-contaminated asphalt that contains even higher concentrations of PAHs (Hugener et al 2004).

In-situ (in-place) vs ex-situ (in-plant)

Although there are some techniques for in-situ (or ‘in-place’) recycling of road pavement, such as remixing and repaving, most asphalt recycling is done ‘in-plant’. This often involves long-distance transport of excavated roading materials, which can create considerable costs and environmental effects, which, from a life-cycle assessment perspective, may be considered objectionable (Karlsson and Isacson 2006).

In-situ remixing and repaving (collectively referred to as ‘surface recycling’) is when a relatively thin layer of an asphalt road is milled or scarified, after which new material (bitumen/aggregate) is added and mixed before laying and compacting the recycled asphalt (Karlsson and Isacson 2006). The difference between remixing and repaving is that the former has new material added, whereas the latter does not. In both cases, the recycled asphalt seal layer is generally overlaid with virgin asphalt. The advantage of in-situ remixing is that there are ‘cold’ and ‘warm’ technologies which circumvent the need for high temperatures, thus decreasing the exposure risk of road-working personnel to tar fumes.

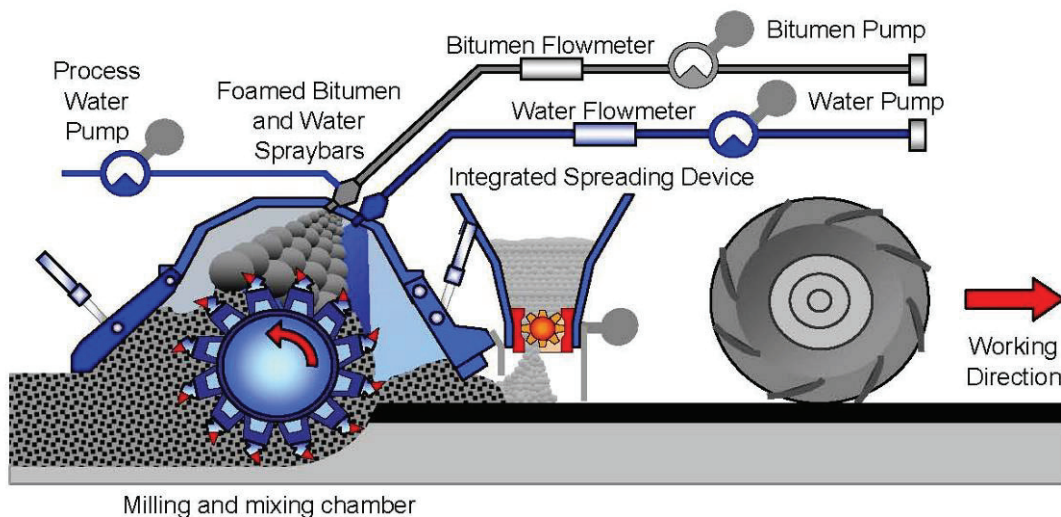
1.2.5.2 Reusing tar-contaminated material as stabilised road base

General principles of stabilisation

The recycling of existing road materials to form the base of the newly reconstructed road is widely used because of significant cost savings and because of the environmental benefits of not having to import or export the large quantities of road materials required for conventional road reconstruction (Lacey 2000). The normal process involves grinding the existing material and adding stabilising agents to improve the mechanical properties of the compacted material. The recycling and stabilisation of a road during reconstruction is typically carried out using in-situ methods.

The two main stabilising additives used are cement (and/or lime) and bitumen (added in either an emulsion or foamed state), the former providing good strength but low crack resistance, and the latter providing less strength but retention of elastic/viscous properties in the compacted layer. In order to obtain the benefits of both types of stabilisers (high strength and flexibility), a combination of bitumen and cement is often employed to achieve the necessary mechanical properties. Emulsion bitumen is a water-based dispersion of bitumen using surfactants, whereas FB is formed by injecting water into hot bitumen, which causes the bitumen to foam as the water is instantly vaporised (figure 1.4). Upon foaming, the bitumen expands to up to 15 times its original volume, facilitating mixing with the pulverised pavement material. In contrast to emulsified bitumen that coats all particle sizes, FB has an affinity for coating fine particulates less than 75µm in size. The coated fines then act as a mortar that binds the recycled matrix together when compacted.

Figure 1.4 Diagram showing the process of in-situ recycling using foamed-bitumen stabilisation. Black and grey road material represents the seal layer and basecourse, respectively. Integrated spreading device is for applying cement



1.2.6 Examples of bitumen ‘encapsulation’ to immobilise contaminants

The addition of bitumen and cement stabilisers to improve mechanical properties of recycled road bases, such as compaction (reduction in porosity), strength and flexibility (crack resistance) also have obvious benefits for immobilising coal tar contaminants. Blending contaminated material with bituminous binders encapsulates the material with relatively benign bitumen and, when compacted, forms a strong matrix with low void volume that resists water penetration. Consequently, the majority of the contaminated surfaces within the matrix are not subject to leaching and/or weathering processes, and as such, the contaminants can be considered to be largely environmentally inert. However, it is not just sufficient to lock the PAH-contaminated material into a solid matrix. Mulder et al (2001) showed that PAH-contaminated material solidified with hydraulic binders (ie cement) still leach PAHs to a relatively high extent. Leaching was reduced 10-fold by the addition of proprietary additives (not specified) that were capable of forming physico-bonds with PAHs.

The subject of reusing and recycling contaminated soil by incorporation into asphalt (hot and cold mixes) and concrete has been reviewed extensively by Testa (1997). With respect to commercial treatment systems, there are patents covering both hot (Meegoda 1994) and cold (MacNeil and McDowell 1996) processes for recycling/reusing up to 20–25% contaminated soils/particulates in asphalt mixes without compromising structural properties. The California company, Encapco Technologies, uses a patented (Jones et al 1999) bitumen emulsion that is mixed directly into contaminated soils (no added aggregate required). The treated materials can be reused as road base (onsite or offsite), general fill, or for a variety of other construction purposes. Leachate testing using the toxicity characteristic leaching procedure (TCLP) has shown that this technology is effective at immobilising heavy metal contaminants.

Of direct relevance to contaminated road material, in 2000 a four-year project to convert the former Oslo international airport at Fornebu to a residential and commercial area was undertaken (Ellefsen et al 2005). A significant challenge of the development was how to deal with tar-contaminated asphalt and sub-base material excavated from beneath the oldest runways. Laboratory leachate testing using tar-contaminated materials containing up to ca 5800mg/kg showed that stabilisation with 3% emulsified bitumen significantly reduced the mobility of PAHs (Ellefsen et al 2005). The stabilisation effect was greatest for the least water-soluble PAHs, for example, a 33-fold reduction in the leaching of benzo(a)pyrene was observed for stabilised samples (relative to non-stabilised control samples). In total, 20,000 tonnes of tar-contaminated material was stabilised and reused as foundation in new roads at Fornebu. In the absence of stabilisation, it would have been necessary to transport the material to a hazardous disposal site at more than double the cost.

To the best of our knowledge, we are not aware of any examples of where FB stabilisation (either in situ or ex situ) has been employed for the purpose of immobilising tar-derived PAHs.

1.2.7 Stabilisation market in New Zealand

Cement and lime stabilisation has been in practice on state highways and local authority (TLA) roads since the 1960s in New Zealand. The industry has grown over the last 20 years to being the standard practice on the majority of maintenance contracts to improve the shear resistance in the pavement layers. Almost all major capital works projects now involve stabilisation, either in the subgrade zone as a subgrade improvement layer or in the pavement layers to improve the performance by either fully binding the aggregates in the case of strongly cemented sub-base or by modifying the aggregates to improve the

shear capacity in the basecourse. Due to commercial sensitivity it is difficult to estimate cement and lime use in New Zealand.

Some sporadic activities of the FB technology dates back to the 1960s but, due to the machine technology not being well developed, it was not pursued beyond some trial sections. The modern FB technology enabled a large-scale introduction into New Zealand in 2004 and since then has grown exponentially. The main applications of FB in New Zealand are heavily trafficked state highways, urban arterial roads and alpine state highways that experience freeze-thaw cycles.

1.2.8 Implications for management of coal tar-contaminated roads in New Zealand

Based on literature studies, in particular, the immobilisation of PAHs with bitumen emulsion in the Oslo airport redevelopment (Ellefson et al 2005), and the favourable life-cycle analysis costs of reusing tar-containing materials in the construction of new roads (Andersson-Sköld et al 2007), there is considerable scope for reusing tar-contaminated road material in New Zealand. While seemingly useful, technologies such as cold and warm in-place asphalt remixing are not relevant to tar-contaminated roads in New Zealand. This is because unlike in Europe, coal tar binders were not used in the production of asphalt, but rather used as primers and first coat seals in chipseal-type road construction. Chipseal roads have relatively thin paved surfaces, and both the composition of stone aggregate and type of bitumen binder makes these roads unsuitable for asphalt recycling. Although emulsion bitumen seems to be used exclusively for contaminant encapsulation, in New Zealand, FB is the only bitumen-based, in-situ recycling method available. Because a key requirement of the study was to use techniques/processes that could be readily adopted by the industry (ie 'off-the-shelf' technology), in-situ recycling using FB-stabilisation represented the most practical solution for recycling tar-contaminated roads in New Zealand.

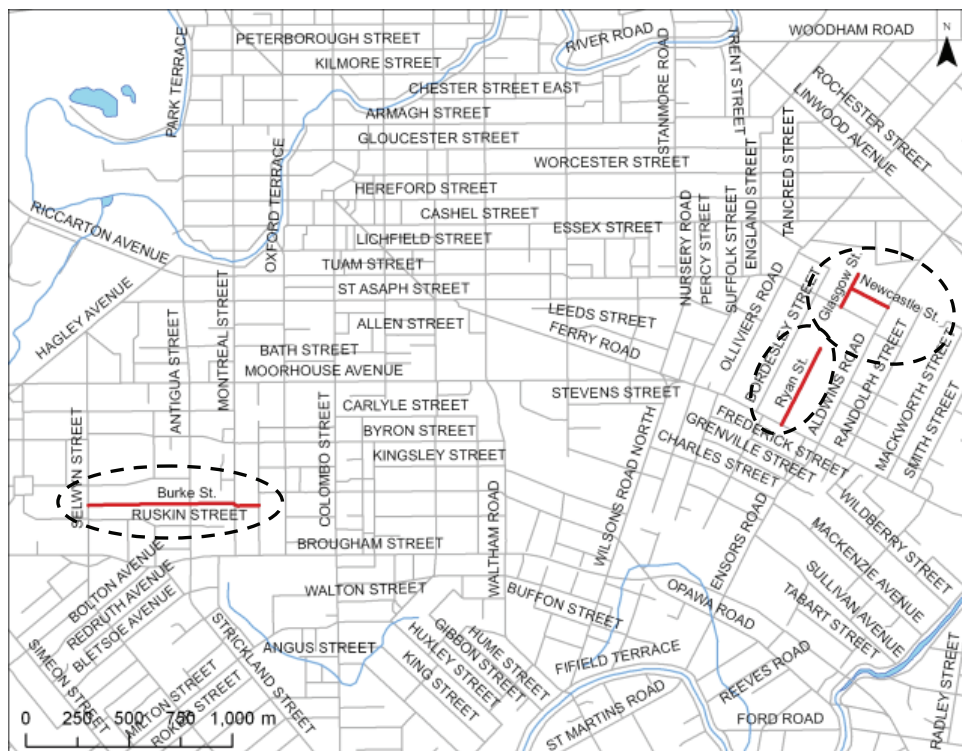
2 Methods

2.1 Site selection

2.1.1 Screening suitability of potential sites identified by Christchurch City Council

Four streets were initially selected by Christchurch City Council as potentially suitable trial sites, namely, Newcastle St, Burke St, Ryan St and Glasgow St (figure 2.1). All four streets were constructed before 1960 and had not been subject to full reconstruction, so coal tar binders used during the original construction of the roads were expected to still be present.

Figure 2.1 Map of Christchurch showing the location of the four streets: Newcastle, Ryan, Burke and Glasgow



Cores (100mm diameter) of the seal layer were taken from four locations along a specified length of each of the four streets. Upon extraction of the core, the thickness of the seal layer was measured and the thickness of the underlying basecourse was estimated. The cores were labelled, wrapped in aluminium foil, and transported back to the City Care depot where they were cut in half down the vertical plane of the core. The sliced cores were then re-wrapped in foil and sent to NIWA laboratories (Hamilton). The vertical cut face of each core was then photographed (Canon G6 digital camera) and, depending on the depth and structure of the core, a vertical slice (typically 15–20mm in thickness; figure 2.2) of the core was cut using a rock saw. From the 15–20mm slice, a 15–30g representative section was removed and repeatedly extracted with dichloromethane (ca 3 x 40ml), at room temperature. A fourth extraction yielded only a mild brown colour that contained <0.5% of the mass removed in the first three extractions. The total

solvent volume was reduced to 50ml and a 1 ml aliquot of each sample was sent to Hill Laboratories (Hamilton) for analysis of PAHs and total petroleum hydrocarbons (TPH). The mass of aggregate remaining was recorded, and the binder content in the seal layer estimated as the difference in mass.

Figure 2.2 Ryan St sample showing how the core was cut. The two halves are the pieces on the left and right, and the centre piece is the ca 20mm slice taken from the right half. Note the small rectangular piece that has been cut in the centre of the slice – this piece was extracted to determine PAH/TPH concentration



2.1.2 Detailed investigation of Burke St

The site chosen for field trials, Burke St, underwent a detailed assessment of the 250m section (from Antigua St to house no.53) by City Care personnel (10 October 2007). Bore holes of approximately 300x400mm (figure 2.3) were excavated at eight locations along the 250m section of Burke St (locations provided in table 2.1), and the thickness of the seal and underlying construction layers were recorded. Excavated seal and construction (basecourse) aggregate from each bore hole was packaged into labelled bags (a total of ca 100kg of material) and sent to the Fulton-Hogan Hamilton laboratory for preparation of design mixes and laboratory samples for leachate testing. After removal of the seal and construction layers, to expose the subgrade material, scala California bearing ratio (CBR) measurements were made to determine the mechanical strength and hence load-bearing capacity of the subgrade material.

Figure 2.3 Bore hole sampling of Burke St. Removal of seal and construction material (left) and undertaking CBR measurement on exposed subgrade (right)



Table 2.1 Location of the eight bore holes taken along the 250m section of Burke St

Bore hole no.	Location
1	chainage 31m, outside house no.88, 2.5m from centre line (CL)
2	chainage 54m, outside house no.83, 2.0m from CL
3	chainage 95m, outside house no.95, 2.0m from CL
4	chainage 120m, outside house no.71, 1.5m from CL
5	chainage 144m, outside house no.66, 2.0m from CL
6	chainage 156m, outside house no.63, 1.5m from CL
7	chainage 190m, outside house no.58, 2.0 from CL
8	chainage 213m, outside house no.53, 1.5m from CL

2.1.3 Laboratory stabilised samples: design mixes

Using Burke St road material excavated from the eight bore holes (ca 100kg), the top seal and basecourse layers were composited in a 4:9 ratio (based on mean thicknesses of 40mm of seal and 90mm of basecourse) and stabilised with different percentages of FB using a bench-top FB generator and mixing apparatus. The following samples were prepared by personnel at Fulton Hogan's Waikato laboratory:

1% FB + 1.5% cement stabilised or 1% FB

2% FB + 1.5% cement stabilised or 2% FB

3% FB + 1.5% cement stabilised or 3% FB

4% FB + 1.5% cement stabilised or 4% FB

0% FB + 0% cement (unstabilised control sample)

Samples (approximately 1kg) of stabilised formulations (1–4) were compacted into cylindrical blocks (100mm diameter by ca 60–65mm high) via the Marshall Automatic Compaction, Type 2 method (ASTMD6926-04). The blocks were then tested for indirect tensile strength using standard methods

(AASHTO – T198). Loose (ie non-compacted) material (ca 3kg) of samples 1–5 were sent to NIWA (Hamilton) for contaminant leaching and algal toxicity testing.

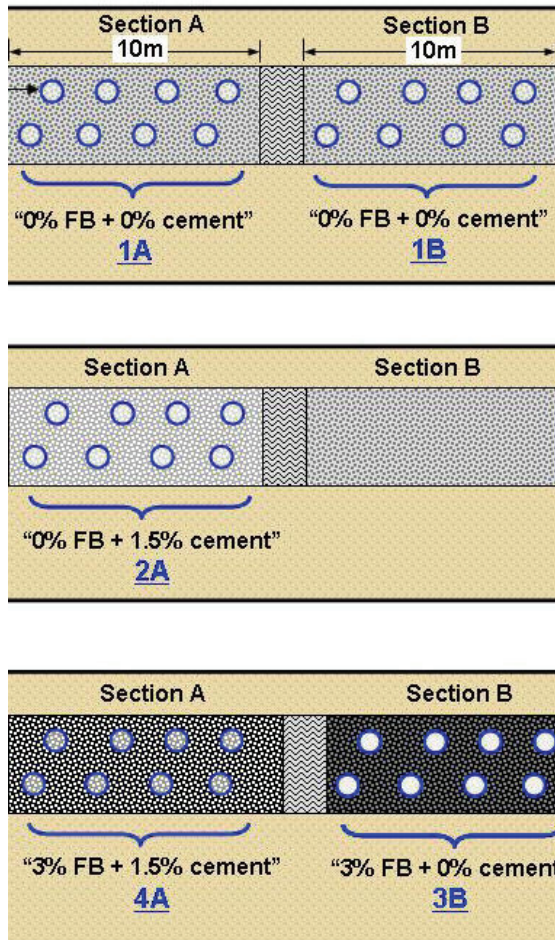
2.1.4 Field trial: Burke Street

The field trial was undertaken on 18 March 2008, using Fulton Hogan’s Wirtgen WR 2500 SK in-situ recycler (figure 2.4, left). An approximately 20m length of Burke St was marked out, consisting of two 10m sections – section A and section B (figure 2.4, right). The process for undertaking the field trial is summarised in figure 2.5, which involved the recycler making a series of passes over the two sections, with five samples being prepared and collected: 1A = unstabilised control material (section A); 1B = unstabilised control material (section B); 2A = cement only stabilisation (section A); 3B = FB only stabilisation (section B); and 4A = FB and cement stabilisation (section A). Each sample was a composite of eight to 10 separate samples (collected with a shovel) along the length of the 10m section. The samples were labelled and then sent to Fulton Hogan Waikato Laboratory for processing.

Figure 2.4 Fulton Hogan Wirtgen 2500 recycler – in-situ recycling train consists of recycler, bitumen tanker and water tanker (left). Section of Burke St used for the stabilisation trial and the approximate locations of test section A and B (right)



Figure 2.5 Diagram showing the process of generating the five samples (including two controls). The circles indicate approximate locations of subsamples that were pooled to give a ca 150kg composite for each of the five sample types



Pass no.1: mill section A and B (no stabilising material added) and material sampled from section A (1A) and section B (1B) for use as non-stabilised control material

Pass no.2: 1.5% cement added to section A (nothing added to section B) and material sampled from section A (2A; 'cement only' stabilisation)

Pass no.3: 3% FB added to section A and B. Material sampled from section A (4A; 'FB and cement'); and section B (3B; 'FB only')

2.1.5 Processing field-collected samples

The five samples were sieved using a 19mm screen; the >19mm fraction was then broken up until all the material passed through a 19mm sieve. The samples were then thoroughly mixed and ca 1kg quantities removed and compacted using the modified Marshall automatic compaction, Type 2 method (ASTMD6926-04) to produce 100mm (diameter) x 60–65mm (high) cylindrical test samples. Three Marshall compacted solids of each stabilised material (ie 2A, 3B and 4A) and the remaining loose material of samples 1A, 1B, 2A, 3B and 4A were sent to NIWA (Hamilton laboratory) for further testing.

At the NIWA Hamilton laboratory, the loose material (ca 100kg of each sample) was subsampled (ca 3kg) using a plastic scoop. The 3kg subsamples were size fractionated using 8mm, 4mm, 1.7mm and 0.6mm sieves to yield five particle size fractions, namely <0.6mm, 0.6–1.7mm, 1.7–4.0mm, 4.0–8.0mm and >8.0mm (figure 2.6). Based on the percent weight contribution of the individual size fractions, duplicate 1kg composites (leaching tests) and triplicate 200g composites (contaminant analysis) for each of the five samples were prepared. The purpose of size fractionating the sample and re-compositing was to reduce the likelihood of obtaining non-representative subsamples from the heterogeneous matrices.

The 200g composites, which had a moisture content of 6–7%, were oven dried at 40°C and exhaustively extracted with dichloromethane at room temperature using periodic ultrasonication. The extracts were made up to a final volume of ca 275ml, at which point 1ml was removed and sent to Hill Laboratories (Hamilton) for analysis of TPH and PAHs (section 2.1.7). The 1kg samples of 'loose' material were used for leaching experiments (section 2.1.6).

Figure 2.6 Composite sample prepared from individual particle size fractions

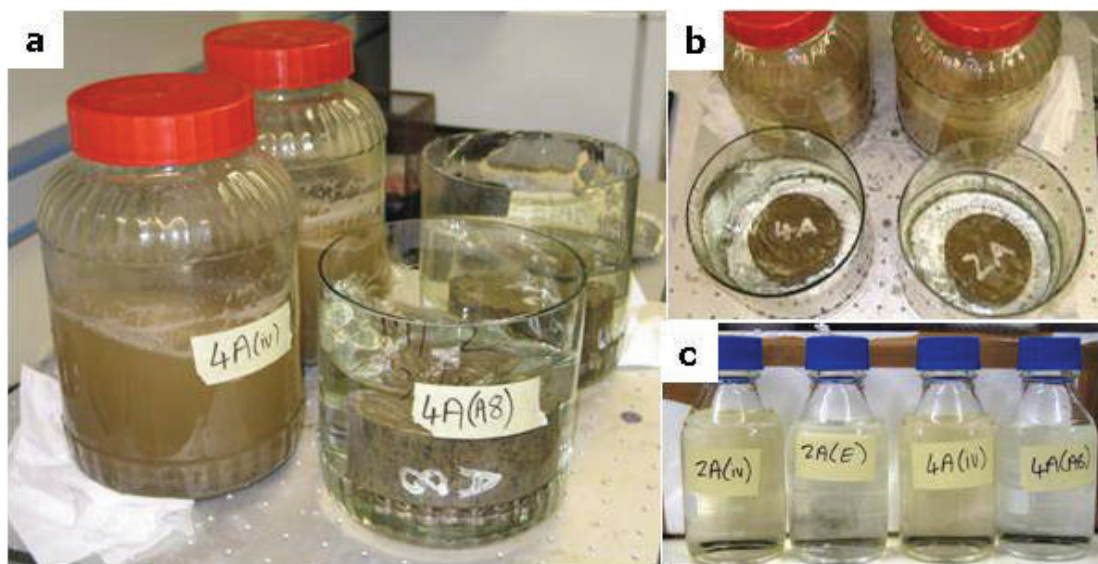


2.1.6 Leaching procedure

The method used for batch leaching was based on the Danish EPA batch leaching test for organic compounds in contaminated soils (Hansen et al 2004). Similar methods were used by Ellefsen et al (2005) to evaluate reduced PAH leaching from bitumen-stabilised tar-contaminated material in both loose and compacted forms. Although there are more common procedures for leaching, such as the toxicity leaching characteristic procedure, TCLP (Method 1311), the purpose of this test is to mimic the acidic conditions of landfill leachates. Because the intended application involves reuse in non-acidic conditions (more likely to be actually alkaline because of cement addition), the TCLP method was not considered relevant. The US EPA also has a synthetic precipitation leaching procedure, SPLP (Method 1312); however, this also employs acidic (non-buffered) extracting solutions that mimic the pH of acidic rain, and uses the same relatively high liquid-to-solids ratio (L/S) of 20.

Briefly, samples (carried out in duplicate) of loose material (1 kg) were added to 5L jars to which 2L of a 5mmol solution of CaCl_2 was added (figures 2.7a and b). The jars containing the samples were shaken gently on an orbital shaker table for 22–24 hours and were left to settle for up to six hours. The majority of the leachate was then decanted off and filtered through a glass fibre filter with a nominal retention of $>1.2\mu\text{m}$ (Whatman GFC). For samples 2A, 3B and 4A, duplicate extractions using compacted cylindrical blocks (1.1kg, prepared by Fulton Hogan) were leached in the same way (5L jar + 2L extraction volume of solvent). After the extraction the intact block was removed and the solution filtered using a $1.2\mu\text{m}$ glass fibre filter. The filtered leachates (figure 2.7c) were sent to Hill Laboratories (Hamilton) for analysis of metals, dissolved organic carbon (DOC), PAHs and TPH. The concentrations of TPH were primarily measured to confirm the presence (or absence) of added FB from the stabilisation process.

Figure 2.7 Leaching of loose and compacted stabilised materials (a and b) and filtered leachates used for chemical analysis and algal toxicity testing (c)



2.1.7 Contaminant analysis

Organic and metal contaminant analyses were carried out using standard methods by an International Accreditation New Zealand (IANZ) accredited laboratory (Hill Laboratories, Hamilton). The standard suite of 16 PAHs listed as priority contaminants by the US EPA were analysed by gas chromatography-mass spectrometry (GC-MS) in selected ion monitoring (SIM) mode, following US EPA Method 8270C. Samples were analysed for TPH concentrations by gas chromatography – flame ionisation detection (GC-FID). This method characterises the concentrations of hydrocarbons in each of a number of different carbon chain length ranges (C_7 – C_9 , C_{10} – C_{14} , C_{15} – C_{36}) and in total (C_7 – C_{36}). The suite of seven heavy metals (arsenic, cadmium, chromium, copper, lead, nickel and zinc) were analysed using inductively coupled plasma-mass spectrometry (ICP-MS).

2.1.8 Leachate toxicity testing

Chronic (ie long-term) toxicity testing of all leachate (from ‘loose’ and ‘compacted’ samples) was determined by the Ecotoxicology Laboratory, NIWA (Hamilton) using the fresh water alga *Psuedokirchneriella subcapitata* (formerly known as *Selenastrum capricornutum*), following NIWA freshwater algae microplate method #15.2. The testing is described as chronic (as opposed to acute or short term) because the test organism undergoes many reproductive cycles (the assay is conducted with diatoms in their exponential growth phase) over the 72-hour duration of the assay. Toxicity is evaluated by comparing algal growth in samples, relative to the growth in controls – increased and decreased growth representing stimulation and inhibition (ie toxicity), respectively. A commonly used value to indicate toxicity is the 50% effects level, or EC_{50} , which in this assay represents the concentration (ie of original leachate) at which a 50% reduction in algal growth, relative to the control/s, is observed.

Prior to algal toxicity testing, the pH of the filtered leachates was adjusted to a neutral pH value (ca 6.8–7.0) that would not be toxic to the algae (the required pH is between ca 6–8.5). Although changing the pH can influence the solubility of heavy metals, this was not considered problematic because the samples had

been filtered so lowering the pH would not result in solubilisation of heavy metals from a particulate phase (with the exception of very fine particulates $<1.2\mu\text{m}$ in size), and thirdly, there were no obvious changes in the leachates (turbidity, colour, etc) to suggest any precipitation or complexation upon neutralisation of leachate.

Using the freshwater leachates (section 2.1.6) a 50% dilution series, consisting of 10 concentrations ranging from 100% through to 0.2%, was prepared for each leachate. The dilutions were dispensed into a microplate (five replicates/dilution), and then algal culture at a concentration of 10,000 cells/ml was added to each dilution. After 72 hours of incubation at 24°C, the concentration of algae was measured using flow cytometry. Algal growth was measured as an increase in the concentration of algal cells. Algal growth tests are considered chronic tests because the effects are assessed over several generations during the exposure period. Toxicity data was analysed using ToxCalc™ statistical software and used to generate EC₅₀ values.

3 Results and discussion

Stage 1: site selection and laboratory trials

3.1 Site selection

There were three criteria for selecting a street: 1) presence of coal tar, 2) presence of sufficient construction thickness to enable in-situ recycling, and 3) on the schedule for reconstruction by Christchurch City Council. While criteria (1) and (3) were considered reasonably straightforward, criterion (2) was more problematic, as relatively thin construction layers (ie seal + basecourse) are quite typical of older city streets in Christchurch. However, if a tar-contaminated streets lacks sufficient construction layer depth, this does not preclude in-situ recycling. In such instances, it is common to apply additional make up aggregate to the road surface to provide the necessary construction layer depth.

Four potential streets were identified by Christchurch City Council that had construction histories consistent with coal tar still being present – these were Newcastle St, Glasgow St, Ryan St and Burke St. Cores (100mm) were taken from four locations at each of the streets, and the amount and uniformity of tar contamination in the respective seal layers assessed by measurement of PAH concentration.

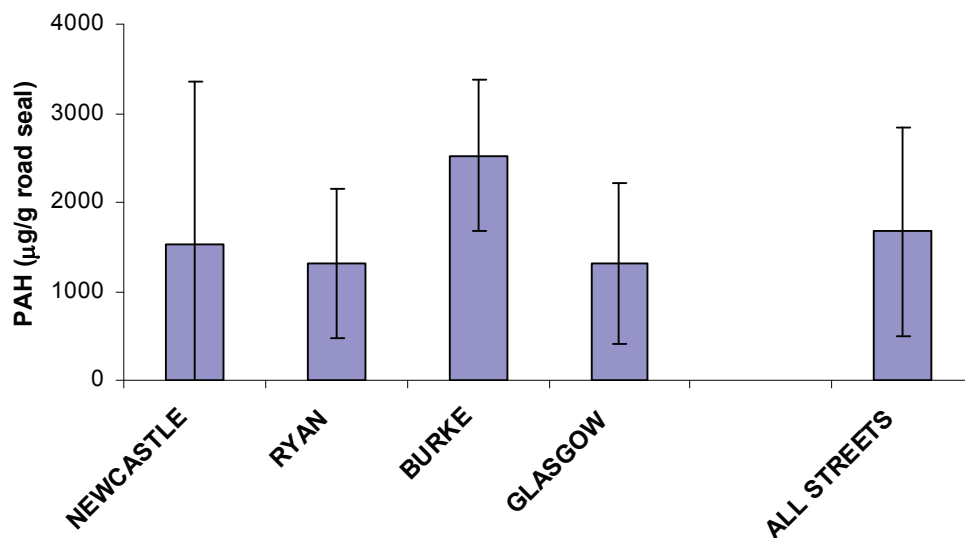
3.1.1 Concentration of PAHs

The mean concentration of PAHs in the seal layer for each street was: Newcastle = 1530 ± 1840 mg/kg; Ryan = 1320 ± 840 mg/kg; Burke = 2520 ± 850 mg/kg; and Glasgow = 1320 ± 900 mg/kg (Figure 12). Burke St not only had the highest mean concentration of PAHs, but also the most uniform distribution of PAH concentrations across the four sites, varying only by a factor of 2.

The concentration of PAHs from these four streets was lower than that found in a previous study of tar-contaminated streets in Christchurch, which reported a mean concentration of 4200mg/kg for five streets (Depree 2005). Literature reports of actual PAH concentrations in tar-contaminated road materials are relatively sparse; however, the observation that 11 of the 16 road samples exceeded the Swiss asphalt recycling maximum of 1000mg/kg of PAHs provides an indication of the relatively high levels present in New Zealand streets. Kreich et al (1999) reported PAH concentrations of between 68 and 10,340mg/kg from cores taken from old Danish mastic and pulver seal layers.

It is also worth mentioning that, with respect to potential toxicity of this material, 15 of the 16 road cores had PAH concentrations that exceeded the PAH toxicity value (LC₅₀) value of 270–340mg/kg for the soil-dwelling insect *F. candida* (dashed red line, figure 3.1). As such, the reuse of this type of material must be carefully managed, and hence the interest in its reuse as stabilised road base to isolated the materials in an ecologically irrelevant compartment. Individual PAH concentrations for the 16 road cores are given in appendix A (table A.1).

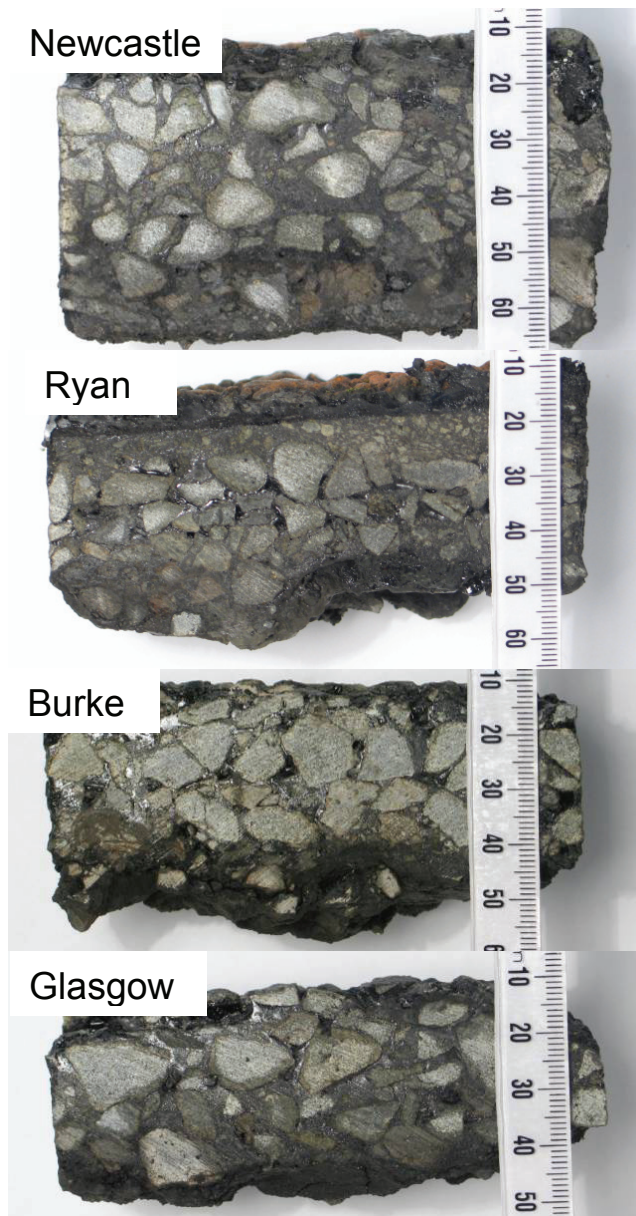
Figure 3.1 Mean road seal PAH concentrations from Newcastle St (N), Ryan St (R), Burke St (B) and Glasgow St (G) - error bars represent 1 standard deviation. Dashed line indicates LC₅₀ value for soil insect, *F. candida* (Deprez 2007)



3.1.2 Seal layer characterisation

The mean thickness (± 1 standard deviation) of the chip seal layer at each street was Newcastle 41 ± 17 mm, Ryan 38 ± 9 mm, Burke 29 ± 5 mm and Glasgow 40 ± 9 mm. The amount of binder was calculated from mass balance of the remaining aggregate after extraction with dichloromethane. Based on this method, the mean binder content (± 1 standard deviation) for each street was Newcastle $10.3 \pm 2.3\%$, Ryan $9.8 \pm 1.3\%$, Burke $7.7 \pm 1.3\%$ and Glasgow $9.3 \pm 1.7\%$. Cross-sectional photographs of representative seal layers for each of the four streets are presented in figure 3.2.

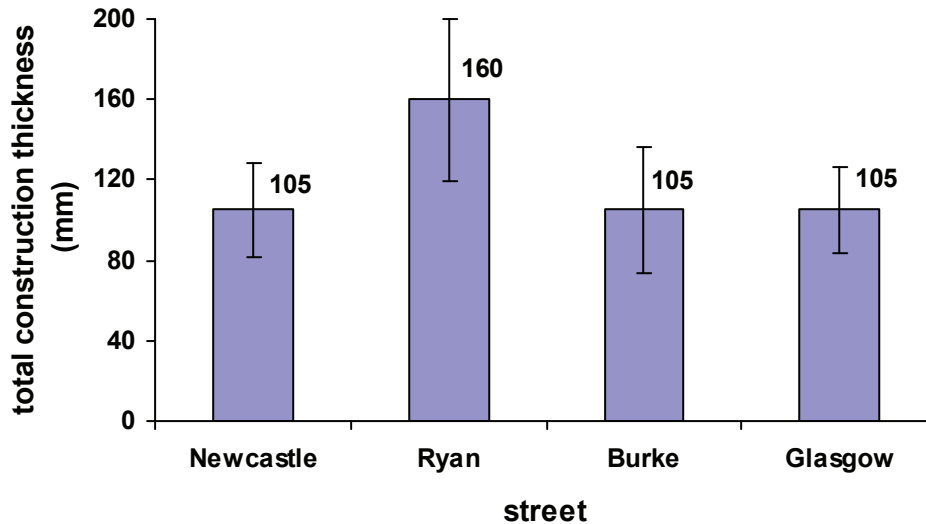
Figure 3.2 Cross-sectional photographs of the seal layer from Newcastle St, Ryan St, Burke St and Glasgow St



3.1.3 Construction layer thickness (seal and basecourse)

Although most of the tar contamination is present in the seal layer, the important parameter for assessing the suitability of a particular street for in-situ recycling is the total thickness of construction material available - this includes the seal layer and the thickness of underlying aggregate/stone referred to as basecourse. Below the basecourse is the subgrade, which generally consists of natural soil or sand. In order for the stabilised recycled material to have sufficient mechanical strength as a road base, a sufficient thickness of construction material is required. The thickness of the recycled layer cannot be increased by incorporating underlying subgrade material because these fine particulates result in inferior, or even unsatisfactory, road base materials. Therefore, to increase the thickness of the recycled layer, it is necessary to apply make-up aggregate prior to in-situ recycling.

Figure 3.3 Mean thickness of construction layers (seal + basecourse) – error bars represent one standard deviation



The mean thickness of the available recycling depth (seal + construction) was 105mm at Newcastle, Burke and Glasgow, and 160mm at Ryan St (figure 3.3). Although the 160mm thick construction depth at Ryan St made it the most suitable candidate for in-situ recycling, the similar construction depths at Newcastle, Burke and Glasgow of ca 105mm indicated that these streets with a thinner construction layer might be more representative of the ‘typical’ tar-contaminated streets requiring reconstruction in New Zealand.

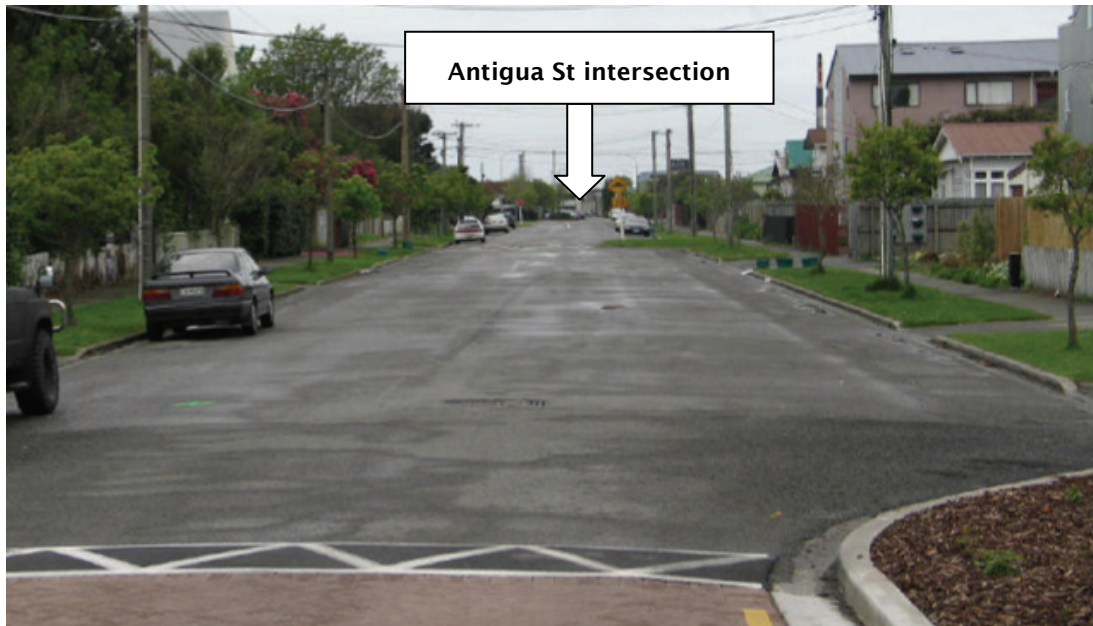
The required recycling depth for FB-stabilisation is dependent on traffic volume, traffic type, design life, and importantly, condition (ie strength) of the subgrade material. Assuming the subgrade properties of a particular road are adequate, for a road with an annual average daily traffic (AADT) volume of 500 and >25-year design life, a FB/cement stabilised base of 130mm would be required. This value suggests that tar-contaminated roads like Newcastle, Burke and Glasgow, with only 105mm of construction, are probably not suitable for in-situ recycling unless additional (ie 30–50mm) make-up aggregate is applied

3.2 Burke St – selected trial site

Based on the initial criteria for selecting a suitable site, it was decided that Burke St represented the best option for evaluating the potential merit of in-situ recycling for reconstructing tar-contaminated streets. Burke St was selected because it had the most uniform tar contamination (based on the concentration of PAHs) and although it had marginal construction thickness, this thin construction seemed to be typical of tar-contaminated roads, and therefore more relevant as a ‘test case.’

A 250m section of Burke St running west from the Antigua St intersection was selected for the FB/cement stabilisation trial (figure 3.4). To fully assess the suitability of this road section, subgrade testing (via CBR testing and Benkelman beam) and design mix investigations for optimal stabilisation were carried out. As the focus of this research report is on contaminant characterisation (concentration, mobility and toxicity), the engineering aspects are discussed only briefly in this section – the complete ‘pavement recycling alternative construction proposal’ completed by Fulton Hogan is provided in appendix B.

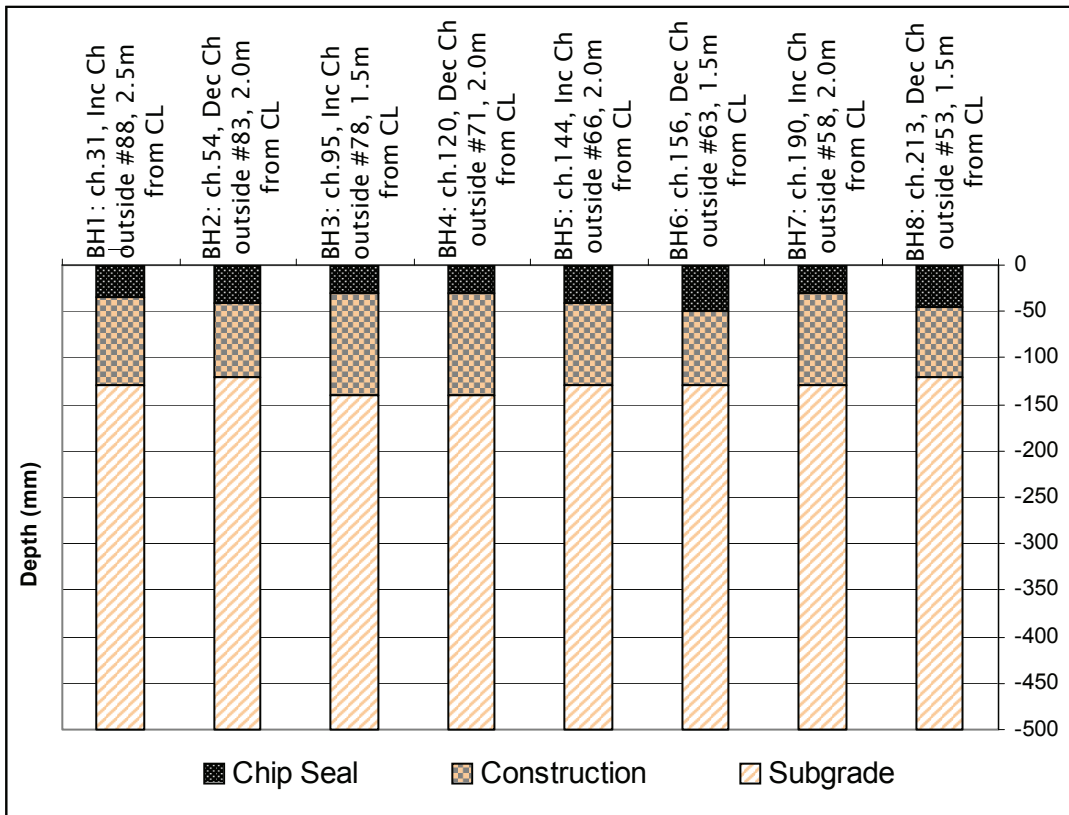
Figure 3.4 The 250m section of Burke St selected for trial – sections runs from the Antigua St intersection (background) to the paved speed bump in the immediate foreground.



3.2.1 Additional characterisation: bore hole excavation

Eight bore holes were excavated along the 250m section of Burke St (refer to chapter 2, table 2.1). At each bore hole (300 x 400mm), the depth of chip seal and construction layers were determined. Across the eight sites, the thickness of seal ranged from 30–50mm (mean 37mm) and the thickness of the basecourse ranged from 80–110mm (mean 92mm). Thus the total construction thickness of Burke St ranged from 120–140mm with an average of 130mm (figure 3.5), compared with only 105mm estimated from taking seal cores (ref to figure 3.3). This was promising because, depending on subgrade quality and traffic parameters, there was potentially sufficient construction thickness without the need for make-up aggregate.

Figure 3.5 Schematic of the bore hole investigations of the existing Burke St pavement



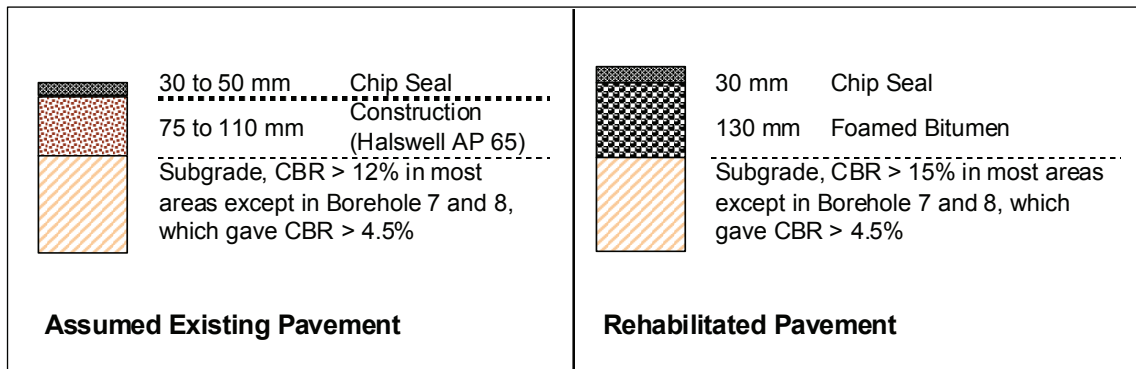
3.2.2 Laboratory mix designs for Burke St stabilisation

To determine the optimal stabilisation treatment for Burke St, Fulton Hogan laboratory personnel prepared and tested a series of FB + cement stabilised test mixes using pavement seal and basecourse material removed from the eight bore holes (results given in appendix B). Based on the mean seal and basecourse thicknesses of 40mm and 90mm, respectively, the bulk Burke St material was recomposed in a 4:9 (seal:basecourse) ratio. The stabilised test mixes prepared were:

- 1 1% FB + 1.5% cement stabilised or 1% FB
- 2 2% FB + 1.5% cement stabilised or 2% FB
- 3 3% FB + 1.5% cement stabilised or 3% FB
- 4 4% FB + 1.5% cement stabilised or 4% FB.

Compacted (via Marshall method) forms of stabilised samples 1–4 were subjected to indirect tensile strength testing, and on the basis of this data, the recommended design mix for Burke St was a 130mm deep recycled pavement stabilised with 3% FB and 1.5% cement. The existing and proposed pavement structures are summarised in figure 3.6. It was noted that the subgrade of Burke St, particularly at bore holes 7 and 8, was not ideal for FB stabilisation and posed a high risk of early failure (Fulton Hogan pavement proposal in appendix B).

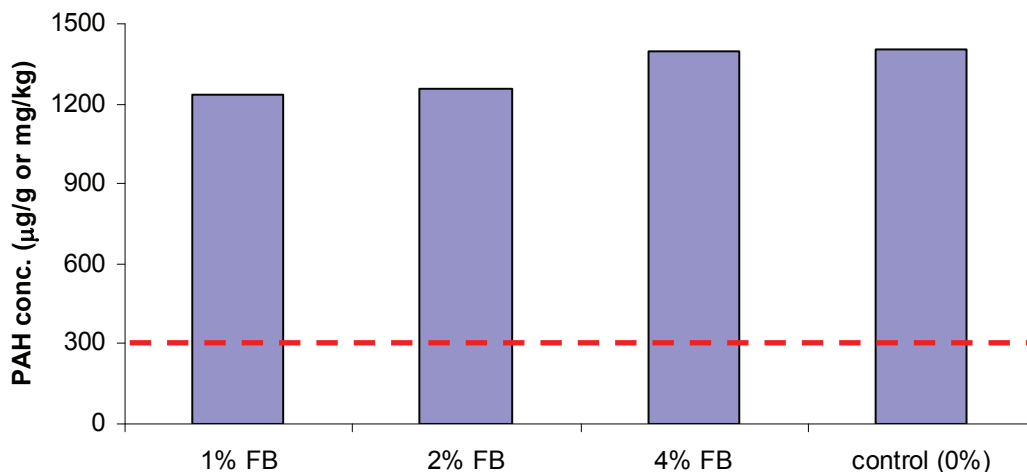
Figure 3.6 Burke St pavement structures – before and after rehabilitation



3.2.3 Concentration of PAHs in mix design samples

In order to compare contaminant leaching between unstabilised (control) and different stabilised formulations (ie test mixes 1-4), the formulations ideally should have similar particulate concentrations of PAHs. To determine whether the bulk of the Burke St material had homogenous concentrations of PAHs, 200g samples of the test mixes containing 1%, 2% and 4% FB (all with 1.5% cement) and unstabilised control material were extracted with dichloromethane and analysed for PAHs. The PAH concentrations in all samples were very similar, ranging from 1200-1400mg/kg (figure 3.7). Individual PAH concentrations are given in appendix A (table A.2).

Figure 3.7 Concentration of PAHs in the Burke St design mixes representative of a ca 130mm recycling depth. Concentrations have not been corrected for stabiliser mass. Red dashed line represents toxic LC₅₀ value for soil dwelling insect *F. candida* (Depree 2007).



Assuming that the majority of the PAHs are from the seal layer (30% of total construction thickness), 1200-1400mg/kg over 130mm depth corresponded to 3900-4500mg/kg of PAHs in the seal layer. This is considerably higher than the mean PAH concentration of ca 2400mg/kg determined from the initial cores (refer to figure 3.1), and more consistent with the median of 4200mg/kg of PAHs previously reported for tar-containing roads in Christchurch (Depree 2006). The low concentrations in figure 3.1 possibly reflect

sampling differences, in particular, whether the brittle tar-primed aggregate material at the seal/basecourse interface is analysed as part of the seal or part of the basecourse layer.

With respect to potential toxicity of the particulate phase, the concentrations of PAHs in Burke St materials (0–130mm depth) were well above the LC₅₀ concentration for *F. candida* (ca 300mg/kg; Depree 2007). Furthermore, using the PAH vs mutagenicity index (MI) correlation of Blackburn et al (1999), the Burke St PAH concentrations correspond to a MI value of ca 6 (refer to figure 1.1) – indicating the materials are carcinogenic. It is acknowledged that this comparison with MI values from the Blackburn et al study is only valid if the composition of PAHs in the tar-contaminated materials is comparable to that of Burke St. While this could not be verified (only PAH totals were reported) the composition of coal tars is expected to be similar because PAH composition is dictated by pyrogenic conditions, which for most coal carbonation processes is ca 1000°C. Furthermore, the comparison is between refined roading grade coal tars, not crude products, which further reduce the likelihood of major compositional differences between coal tars in the two studies.

3.2.4 Contaminant leaching: laboratory prepared mix designs

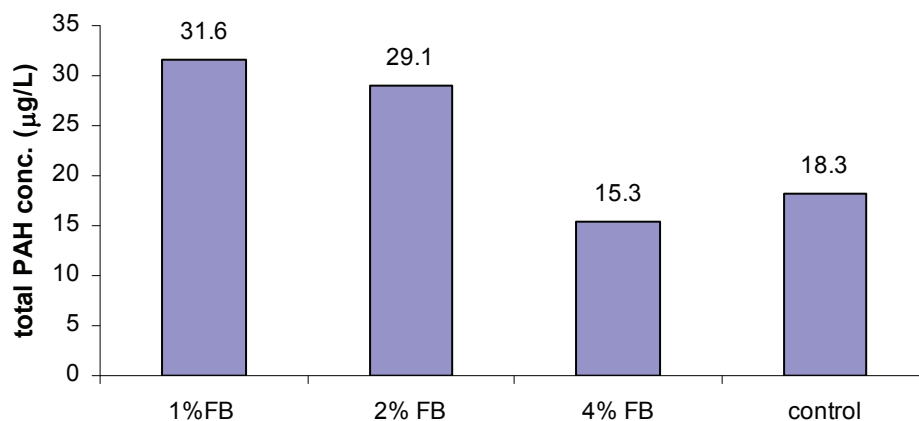
To determine the effect of FB on contaminant leaching, the 1%, 2% and 4% FB-stabilised samples (all with 1.5% cement) and unstabilised control were extracted with water for ca 24h, and the leachates analysed for PAHs, TPH, DOC and heavy metals. Although heavy metals, unlike PAHs, are not associated with coal tar, these were analysed to determine to what degree alkalinity (from 1.5% cement) had an effect on their leaching and to compare the behaviour (ie partitioning tendency to FB) of hydrophilic dissolved metal ions with hydrophobic PAHs. The objective of the preliminary investigation using laboratory-prepared samples was to determine whether the optimal formulation for achieving the required engineering parameters (ie strength) coincided with the formulation that achieved the lowest contaminant bioavailability; or whether optimisation of one compromised the other. The initial intention was to also prepare and test formulations consisting of only FB (ie no cement) in order to access the influence of cement (eg alkaline pH); however, as a result of communication problems these formulations were not provided by Fulton Hogan laboratory personnel. Furthermore, the aim was to compare the difference between loose and compacted stabilised samples; however, compacted solid samples of 1–4 were not provided by the Fulton Hogan laboratory. As such, the preliminary testing for contaminant immobilisation was performed only on loose sample matrices.

3.2.4.1 Polycyclic aromatic hydrocarbons (PAHs)

The total concentration of PAHs in Burke St leachates ranged from 15 to 32µg/L (figure 3.8, and refer to appendix A – table A.3 for individual PAH concentrations). Although the comparisons are based on single leachate samples (ie no replicates), the 4% FB stabilised sample yielded marginally lower leachate concentrations of PAHs than the unstabilised control (15.3µg/L compared to 18.3µg/L). The 1% and 2% FB-stabilised samples both contained ca 30µg/L of PAHs. It is worth mentioning that phenanthrene, the most dominate PAH in other leachates, was below detection limit in the control sample. This was difficult to reconcile, and may represent an analytical artefact – if so, the total concentration of PAHs in the control sample would have been ca 30µg/L (based on 12µg/L of phenanthrene in the 1% FB leachate). However, if the total PAH concentration for the control was real, a possible reason for the greater PAH concentrations in ‘FB and cement’ leachates, was the increased concentration of DOC (75mg/L compared to 30mg/L) caused by alkaline leaching conditions (pH 11–12). Cho et al (2002) has reported the water solubility of pyrene is increased by factors of ca 3 and 6 in the presence of 10mg/L and 20mg/L, respectively, of natural organic matter (ie aquatic humic/fulvic material). This highlights the importance of comparable

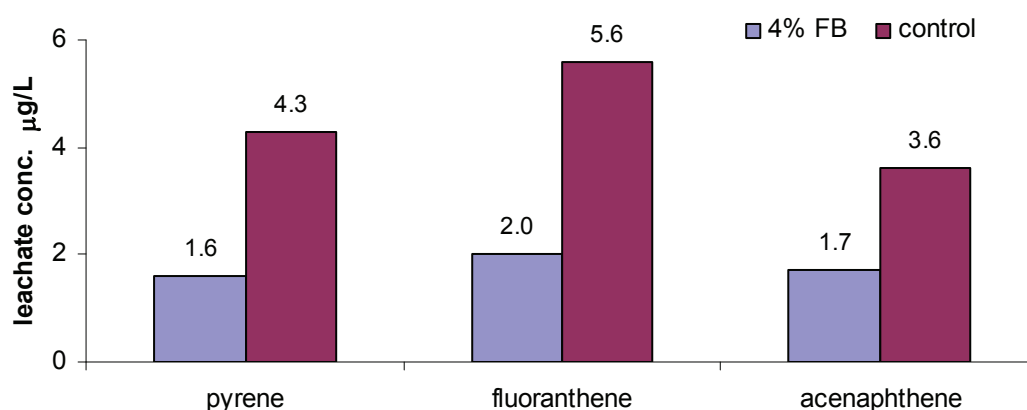
sample types and hence better comparisons would have been possible with i) 'cement only' – to determine the effect of pH on PAH leaching without FB) and ii) 'FB only' – to compare the effect of FB on PAH leaching without interference from strongly alkaline leaching conditions.

Figure 3.8 Leachate concentrations of total PAHs in Burke St samples (all FB samples contained 1.5% w/w cement)



Three- and four-ringed PAHs were the most abundant PAHs with six of these PAHs accounting for over 90% of the total – namely, acenaphthene, anthracene, fluorene, phenanthrene, fluoranthene and pyrene. Relative to the unstabilised material, the 4% FB sample contained two to three times lower individual concentrations of abundant PAHs (ie pyrene, fluoranthene and acenaphthene) than the unstabilised control material (figure 3.9). These results tend to support the notion that the non-detection of phenanthrene in the control leachate was an analytical artefact. The decreased mobilisation of individual PAHs was similar to, but less pronounced than, the 4.7- and 4.2-fold reductions in pyrene and fluoranthene, respectively, reported by Ellefsen et al (2005) in a Norwegian study investigating contaminant leaching from coal tar-contaminated millings that contained up to 5800 µg/g of PAHs stabilised by emulsified bitumen. The lower reductions in PAH mobilisation in the Burke St samples relative to those in the Norwegian study are likely because FB stabilisation mainly coats particles <75µm, whereas emulsion stabilisation used in the Norwegian study coats all particle sizes with bitumen. Relative to emulsion stabilisation, FB will be less effective at encapsulating coal tar material, resulting in a greater proportion of contaminated material available (ie exposed) to leach into the water extractant.

Figure 3.9 Comparison of selected PAH leachate concentrations from 4% FB (+1.5% cement) stabilised and control (no cement or FB) samples from Burke St



3.2.4.2 Heavy metals

The suite of metals analysed included arsenic, cadmium, chromium, lead, nickel, copper and zinc. Of these, copper and zinc tend to be the most toxic to aquatic life. While heavy metals are not a characteristic contaminant in coal tar, they were analysed for two main reasons. First, to provide additional information on contaminant behaviour using a 'hydrophilic' contaminant (ie when present as dissolved metal cations), and second, to determine whether any leachate toxicity is attributable to contaminants other than hydrophobic tar-derived PAHs, which are typically present in water at relatively low concentrations, despite abundance in the particulate phase, because of their hydrophobicity.

The total concentrations of leached heavy metals ranged from 170–180µg/L in stabilised samples (pH ca 12), down to 26µg/L in the control material (pH 6.4). Individual concentrations for the seven heavy metals analysed are provided in table A.4 (appendix A). The alkaline conditions caused by the cement influenced both the total concentration and relative composition of solubilised heavy metals (figure 3.10). Copper and chromium each accounted for ca 40% of the total metal concentration in the 'FB + cement' stabilised samples that had leachate pH values of ca 12. In contrast, zinc accounted for >80% of total metals in the unstabilised control leachate, which had a lower pH value of 6.4. The similar total concentrations for 1, 2 and 4% FB-stabilised samples (ie 170–180µg/L) demonstrated that pH is the most important factor controlling metal solubilisation, with FB content having little, if any, influence.

Figure 3.10 Composition of heavy metals leachates from Burke St samples. FB stabilised samples contained 1.5% cement resulting in leachate pH of 12; 'control' contained no cement (pH 6.4)

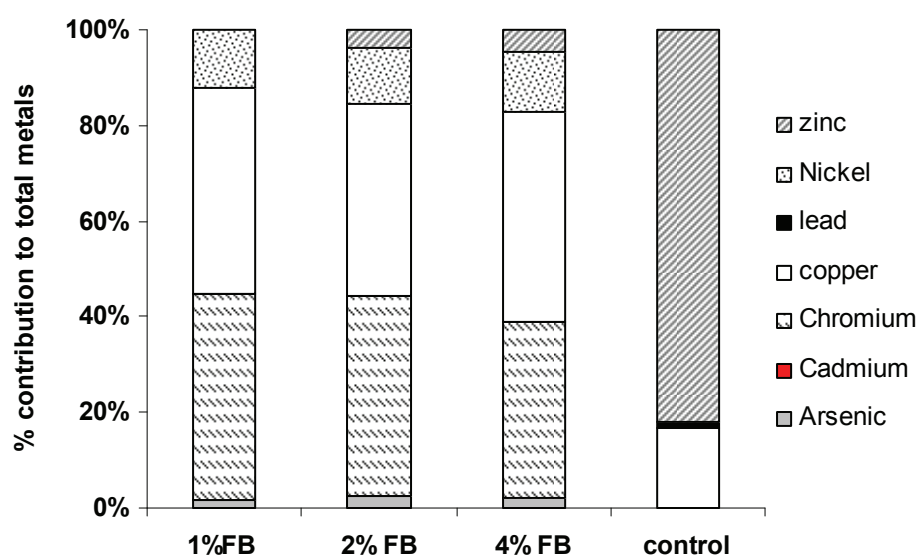


Figure 3.11 Chromium leachate concentrations of Burke St samples – highlighting the increased mobilisation under alkaline conditions (pH 11–12) arising from the cement in FB-stabilised samples

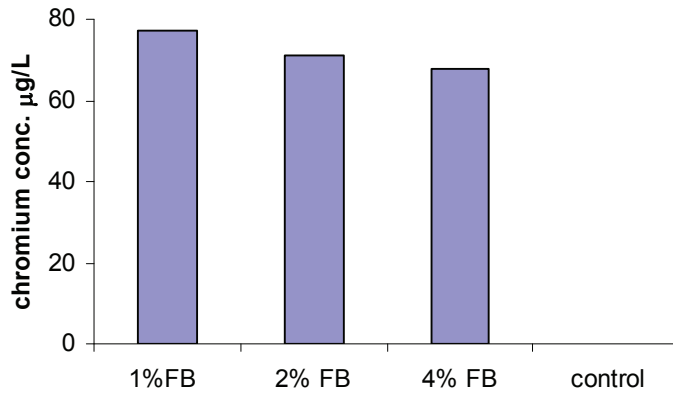
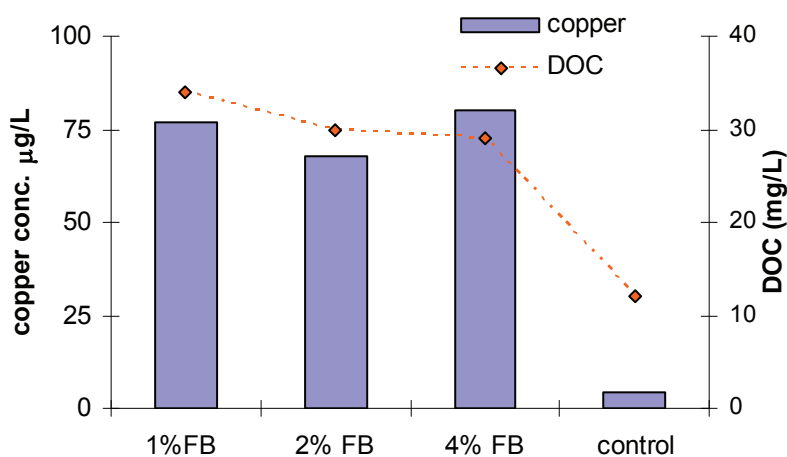


Figure 3.12 Increased leaching of copper in the presence of higher DOC concentrations. Despite lower solubility of Cu(II) at alkaline pH values caused by the presence of cement, complexation of copper with DOC results in enhanced solubility (all FB-stabilised samples also contained 1.5% w/w cement)



Alkalinity has a major effect on the solubility of metal hydroxides with solubility general decreasing with increasing pH. This did not translate into lower concentrations of leached metals in ‘FB + cement’ stabilised samples for two reasons: 1) chromium has an aqueous solubility curve that has a minimum at ca pH 7–9 and then increases at higher pH values (Remoundaki et al 2007), resulting in greater chromium solubilisation (figure 3.11); 2) alkaline pH conditions produced leachates with more metal-binding DOC, increasing the solubilisation of copper (figure 3.12) via DOC-Copper complexation (Impellitteri et al 2002; Jordan et al 1997; Temminghoff et al 1997; Yin et al 2002). The leached copper concentration was not proportional to the increase in DOC presumably because the enhanced solubilisation was offset by the decreased aqueous solubility of copper at pH12 (cf pH 6.4).

3.2.4.3 Comparison with water quality guidelines and toxicity values

To complete the stage 1 evaluation of the potential merits of in-situ stabilisation of tar-contaminated road materials, leachate concentrations were compared to aquatic toxicity values to give a preliminary assessment of potential toxicity. For selected contaminants, the leachate concentrations have been compared with HC₅₀ values (ie concentration of a contaminant that is hazardous to 50% of aquatic species, Haye et al 2007) in table 3.1. Copper was the only heavy metal that exceeded the HC₅₀ value for aquatic species (highlighted in bold, table 3.1), and this was attributed to higher mobilisation via DOC-copper complex formation. An important point is that despite the concentrated nature of the extractions (L/S ratio of 2), the tar-related contaminants of concern, PAHs, were present at relatively low concentrations, for example, fluoranthene concentrations were well below the HC₅₀ value of fluoranthene.

Table 3.1 Algal toxicity (indicative only) parameters of Burke St leachates

Contaminant	Burke St samples (µg/L)				HC ₅₀ ^(a) value
	1% FB	2% FB	4% FB	control	(µg/L)
metals					
chromium ^(b)	77	71	68	<0.5	220
copper	77	68	80	4.2	18
zinc	<11	6.3	8.6	21	89
PAHs					
fluoranthene	4.7	4.1	2.0	5.6	30
pyrene	2.9	3.4	1.6	4.3	-

^(a) HC₅₀ = hazardous concentration; the concentration of a contaminant expected to cause an effect (death/growth inhibition) in 50% of aquatic species [derived from the 50% effects level for individual species]; values taken from Lijzen et al (2001).

^(b) Cr(III) values have been used.

Stage Two: field trial

Although some of the findings from the laboratory-prepared samples were inconclusive (regarding the contaminant immobilisation merits of FB-stabilisation), approval was given to advance to stage 2 (field trial) since stage 1 had shown that there was very low risk from the leaching of PAH contaminants.

3.3 Burke St field in-situ recycling trial

The trial on Burke St was carried out on 18 March 2008. As discussed in section 3.2.2, the recommended rehabilitation for Burke St was a 130mm deep recycled pavement layer stabilised with 3% FB and 1.5% cement. The option to use make-up aggregate was considered unnecessary. The intention was to collect five samples from two 10m sections by having the recycling machine make multiple passes (up to three) over each of the sections (figure 3.13). The sample types are summarised in table 3.2.

Figure 3.13 Wirtgen 2500 in-situ recycler stabilising Burke St with 3% FB and 1.5% cement – direction of travel is toward the left of the picture, with the stabilised material visible behind the rear wheels of the recycler



Table 3.2 Planned control (unstabilised) and stabilised samples from Burke St field trial

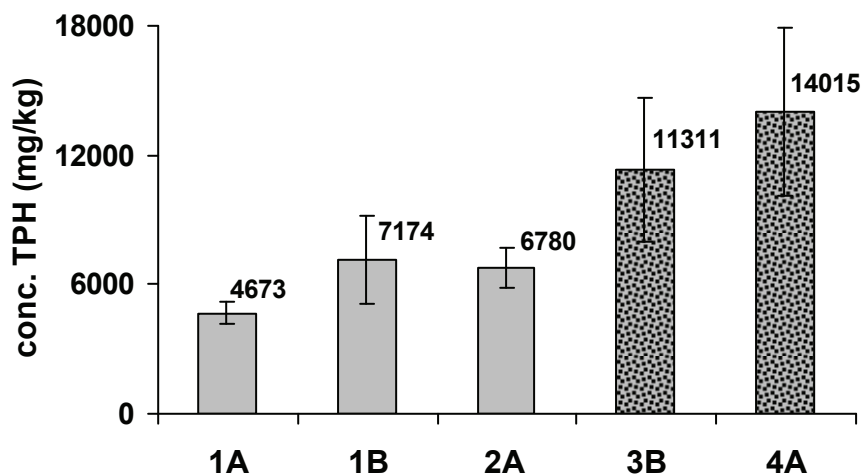
Sample code	Cement (%)	FB (%)	Comment
section A			
1A	0	0	Control sample for section A
2A	1.5	0	Cement only stabilisation
4A	1.5	3	Cement + FB stabilisation
section B			
1B	0	0	Control sample for section B
3B	0	3	FB only stabilisation

While most in-situ FB rehabilitations include cement, the specification of ‘cement only’ and ‘FB only’ samples was to attempt to determine the relative importance of each stabilising component with respect to controlling the leaching of contaminants from the loose and compacted matrices. In preliminary tests performed using laboratory-prepared formulations (section 3.2.4), there was some concern regarding the alkalinity of leachates (pH 11–12) generated by cement-containing stabilised materials. These alkaline conditions resulted in increased concentrations of DOC, and presumably the increased solubilisation (ie mobilisation) of PAHs. To address the uncertainties that carried over from stage 1, ‘bitumen only’ (3B) and ‘cement only’ (2A) samples were included in the field trial.

3.3.1 PAH and TPH concentration in recycled matrices

The mean concentrations of TPH in the five loose samples ranged from ca 5000–7000mg/kg in the samples containing no FB to 11,000–14,000mg/kg for the two samples containing FB (figure 3.14; refer to table A.5, appendix A for individual concentrations). The approximately two-fold higher TPH concentrations in FB-stabilised samples were consistent with the addition of 3% FB to a 130mm thick recycling layer that contained 40mm of chip seal containing 8% binder. However, it is unclear why the absolute TPH values are low, because assuming most binder is soluble in the extraction solvent (dichloromethane), a 1%, 2% and 3% binder content should correspond to ca 10,000, 20,000 and 30,000mg/kg of TPH, respectively. Despite the uncertainty in absolute TPH concentrations, the concentrations in figure 3.14 confirmed the presence of FB in samples 3B and 4A.

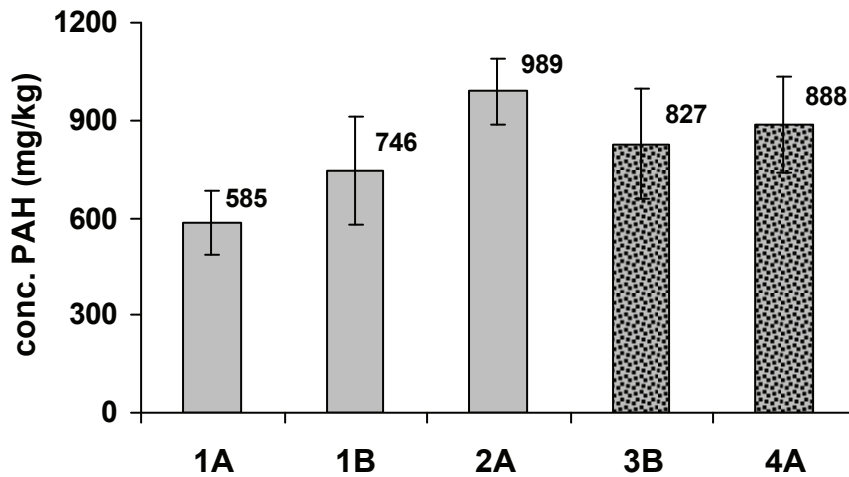
Figure 3.14 Mean concentrations (n=3) of TPH in the field stabilised materials from Burke St. Black stippling shows samples containing FB (3B and 4A), 1A and 1B are controls containing no stabilising agents, and 2A contains cement only (no FB). Error bars represent ± 1 standard deviation



The mean concentration of PAHs in the in-situ recycled materials ranged from ca 600–1000mg/kg (figure 3.15). Individual PAH concentrations are given in table A.5 (appendix A). In contrast to TPH, the addition of 3% FB is insignificant to the amount of PAHs present from coal tar binders because 3% of ca 30mg/kg (concentration of PAHs in bitumen) equates to a PAH contribution of only ca 1mg/kg, which is negligible in light of the elevated PAH concentration of the original material. The PAH concentrations in the field samples were considerably less than the 1200–1400mg/kg observed for the laboratory prepared Burke St design mixes (refer to figure 3.7). Presumably the decreased PAH concentrations reflect either: 1) genuine differences between the section of stabilised road and material sampled earlier from borehole investigations; or 2) an overestimation of the seal layer contribution to total construction thickness, resulting in laboratory-prepared composite that contained too much tar-contaminated binder material.

Despite triplicate analysis, control 1A did not seem to be representative of section A material, given that other samples of section A materials (ie 2A and 4A) contained higher concentrations (900–1000mg/kg). This may have been because the composition of material in section A after one pass of the recycler was different from that after the second (2A) and third (4A) pass.

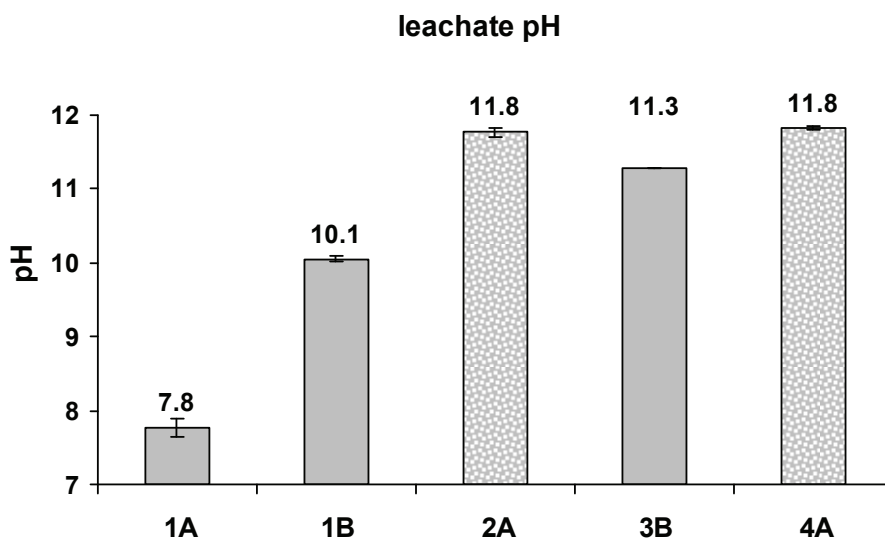
Figure 3.15 Mean concentrations (n=3) of PAHs in the field stabilised materials from Burke St. Black stippling indicates samples containing FB (3B and 4A), 1A and 1B are controls containing no stabilising agents, and 2A contains cement only (no FB). Error bars represent ± 1 standard deviation



3.3.2 pH of leachates: test for presence/absence of cement

The pH of the leachates from all five samples is shown in figure 3.16. The only samples that should have contained 1.5% cement stabilisation were 2A and 4A (highlighted by white stippling on bars); however, it is apparent from the alkaline pH that the 'bitumen only' (3B) and section B control (1B) samples also contained significant amounts of cement (leachate pH of 11.3 and 10.1, respectively). The only sample that did not contain cement was control sample 1A. Confirmation of cement addition would have been provided from indirect tensile strength measurements performed on compacted samples; however, delays in processing the samples resulted in the cement hydrating before compaction.

Figure 3.16 pH of leachate solutions showing that 'cement-free' samples 1B and 3B most likely contained some cement (white stippling indicates those samples that were supposed to contain 1.5% cement, ie 2A and 4A). Error bars represent the difference between duplicate samples



The incorporation of cement in samples 1B and 3B may have been caused by operator error and/or accidental contamination resulting from the relatively small size (10m) of the test sections. Regardless of the reason, the presence of cement in 3B means that it was not possible to assess the immobilisation of PAHs without the interference of increased DOC concentrations caused by alkaline leachates. Although disappointing from a research perspective, the aim was to evaluate a practical solution to tar-contaminated roads, and the reality is that most (if not all) in-situ FB pavement rehabilitations in New Zealand involve cement addition. In this regard, the field-stabilised samples still enabled comparisons between 'FB + cement' (3B and 4A) and a 'cement only' sample (2A). With respect to non-stabilised controls, although 1A was essentially cement free, it did not appear to be representative of the amount of tar contamination present in section A, and so more emphasis is placed on control 1B in the subsequent discussion.

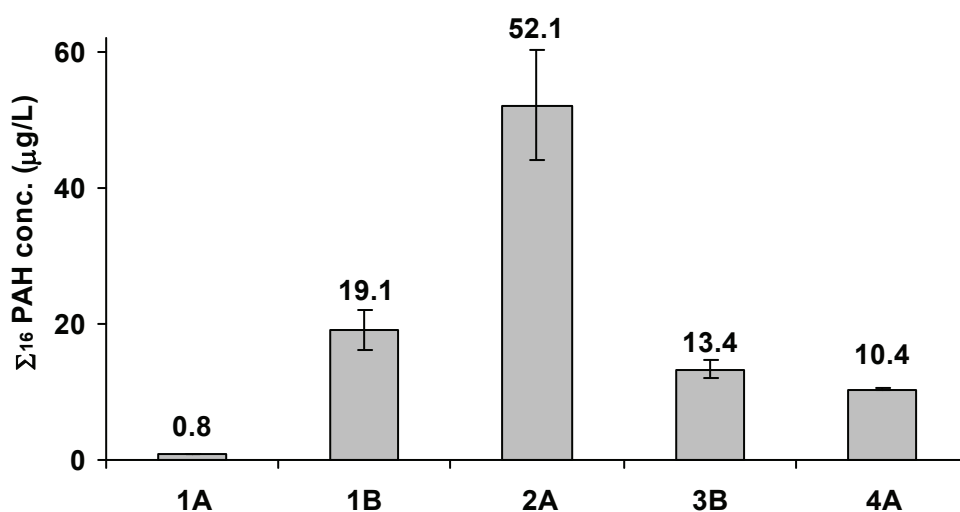
3.3.3 Contaminant leaching: field in-situ stabilised materials

3.3.3.1 PAHs: leaching of non-compacted road materials

The concentrations of PAHs ($\mu\text{g/L}$) leached from the four loose samples are shown in figure 3.17 (refer to table A.6, appendix A for individual PAH concentrations). The total leachate concentrations ranged from $0.8\mu\text{g/L}$ in the control (1A) to $52\mu\text{g/L}$ in the cement only material (2A). In comparison, Karlsson and Viklander (2008) reported dissolved total PAH concentrations in catchpit (ie gully pots) water (pH 7.4) of $34 \pm 60\mu\text{g/L}$.

The very low solubilisation of PAHs in control sample 1A (reproduced in duplicate samples) was difficult to reconcile because it was so much lower than the $18.3\mu\text{g/L}$ of PAHs in the leachate from the laboratory control sample (refer to figure 3.8 in section 3.2.4.1). This value of $18.3\mu\text{g/L}$ was, however, in very close agreement to the field control sample 1B, which contained $19.1\mu\text{g/L}$ of PAHs. The difference between the solubilisation of PAHs in 1A and 1B was not readily explained by differences in DOC concentration because this only varied by a factor of two (6.5 and 12.0mg/L , respectively).

Figure 3.17 Leachate concentrations of PAHs from loose (non-compacted) material. Samples: 1A and 1B - controls (no FB or cement); 2A - cement only; 3B and 4A - 'FB + cement'. Error bars represent the difference between duplicate samples

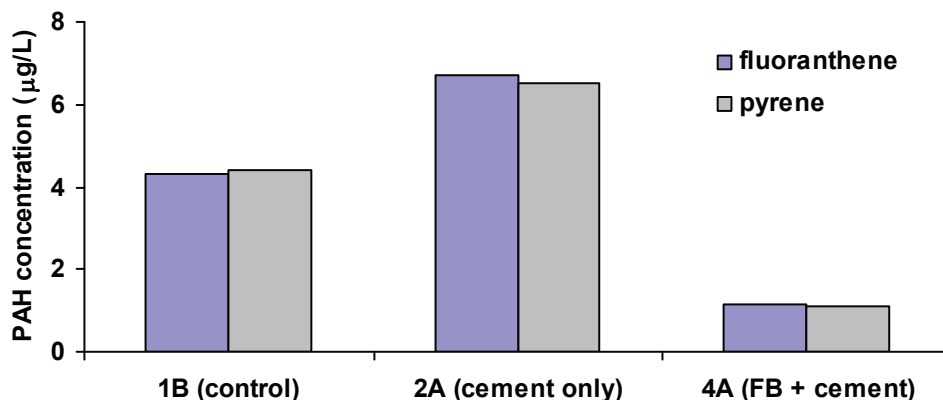


Of the 16 PAHs, seven two- to four-ringed PAHs, namely naphthalene, acenaphthene, fluorene, phenanthrene, anthracene, pyrene and fluoranthene, accounted for >95% of the total PAH. Larger five- and six-ringed PAHs are only sparingly soluble in water and have very large K_{OW} values, which mean these PAHs are strongly bound/partitioned to organic phases (ie road binders). The highest individual PAH concentration was 8.6 $\mu\text{g/L}$ of fluoranthene in the 'cement only' leachate (2A). The results from field-stabilised samples were consistent with the laboratory results (refer to figure 3.8), with the addition of FB decreasing the mobilisation of individual PAHs (figure 3.18) – especially relative to the 'cement only' sample, 2A. For example, relative to the control (1B) and 'cement only' (2A), the fluoranthene concentration in 'cement + FB' (4A) leachate was lower by factors of 4 and 6, respectively.

The addition of cement to the milled roading material resulted in increased leaching of PAHs, with the 1.5% 'cement only' sample (2A) yielding a total PAH leachate concentration of 52 $\mu\text{g/L}$. PAH leaching of 3B (ie supposedly FB only) was very similar to that of 4A (cement + FB), which further supports the assumption that this sample contained a cement loading similar to that of 4A (ie 1.5%).

The most comparable study involved the addition of emulsified bitumen to remediate tar-contaminated soil/aggregate from the redevelopment of Fornebu airport, Oslo, Norway (Ellefsen et al 2005). In this study, contaminated material was excavated, transported to an emulsion bitumen 'batch-type' plant and stabilised with 3–3.5% bitumen, before being used as foundation material for newly constructed roads in the Fornebu urban redevelopment. Under comparable batch leaching conditions (L/S 2.25), relative to non-stabilised samples, Ellefsen et al reported similar reductions of 4.2- and 4.7-fold for fluoranthene and pyrene, respectively.

Figure 3.18 Mean (n=2) leachate concentrations of pyrene and fluoranthene showing decreased mobilisation in FB-stabilised, tar-contaminated road material



3.3.3.2 PAHs: leaching of compacted road materials

An important component of the field testing was to determine the effect of compaction on the leaching of PAHs, since this is the final form of the stabilised material in road base applications. The hypothesis was that while some PAH immobilisation might be attributable to coating/encapsulation, the main mechanism for reducing PAH leaching would be the reduced surface area of contaminated material exposed to leaching when compacted.

Non-stabilised control samples could not be compacted, and so leaching from compacted matrices was limited to 2A, 3B and 4A. Examples of the Marshall-compact cylindrical blocks of stabilised materials used for leachate testing are shown in figure 3.19. The mean PAH leachate concentrations of loose vs compacted

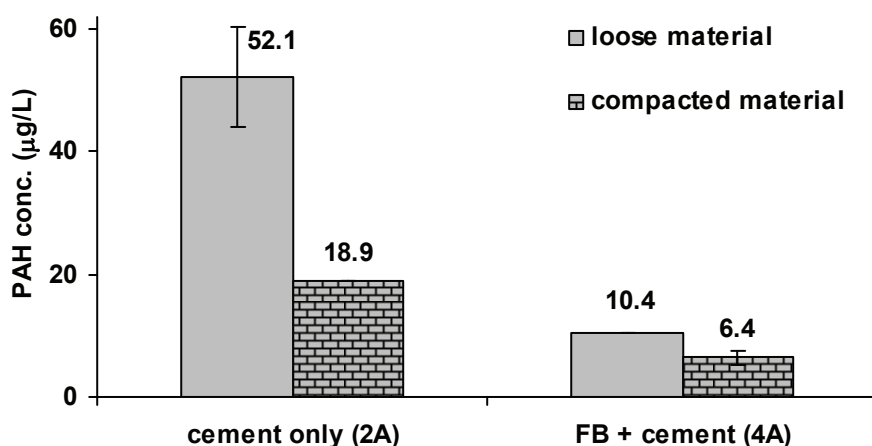
sample matrices for samples 2A and 4A are given in figure 3.20 (refer to table A.7, appendix A for individual PAH concentrations). Compaction of 2A (cement only) resulted in the amount of solubilised PAHs decreasing from a mean of 52µg/L to 19µg/L (2.7-fold decrease). The decrease upon compacting 4A (cement + FB) was less pronounced (ca 1.6-fold), decreasing from a mean of 10.4µg/L to 6.4µg/L. Sample 3B (data not shown), as expected, was similar to 4A, decreasing from 13.4µg/L to 9.1µg/L upon compaction.

Figure 3.19 Compacted samples of 2A (cement only) and 4A (cement + FB) after 24 h leaching test



Comparing directly with relevant samples from the Norwegian study (Ellefsen et al 2005), it was apparent that the absolute concentration of leached PAHs was significantly (five- to seven-fold) greater. For example, a tar-contaminated sample containing 957mg/kg of PAHs and stabilised with 3–3.5% emulsified bitumen resulted in a leachate PAH concentration of 1.3ug/L using a L/S ratio of 2.25 (cf with 2 for Burke St samples). The lower leaching may reflect differences in the emulsified bitumen vs FB process (eg better coating via emulsion application), but another important difference between the two stabilisation protocols was the absence of cement in the Norwegian study. Accordingly, it is assumed that the leachates of that study were not strongly alkaline and hence did not facilitate PAH solubilisation via concomitant increases in DOC.

Figure 3.20 Comparison of leachate concentrations of PAHs from 'loose' (grey bars) and 'compacted' (brick patterned bars) stabilised materials



Water exclusion from the compacted matrices was not optimal because the cement had hydrated prior to the samples being compacted, however water sorption was <1% (by weight) for all compacted samples. As such, a substantial amount of the 1.1 kg of stabilised material within the compacted matrix would not have been exposed to leaching by water. This was supported by the four- to five-fold reductions in the leachate concentrations of copper and chromium (next section) upon compaction, which are considerably less hydrophobic than PAHs, and therefore less affected by water-solid partitioning interactions.

3.3.3.3 Explanation for PAH leaching via equilibrium partitioning

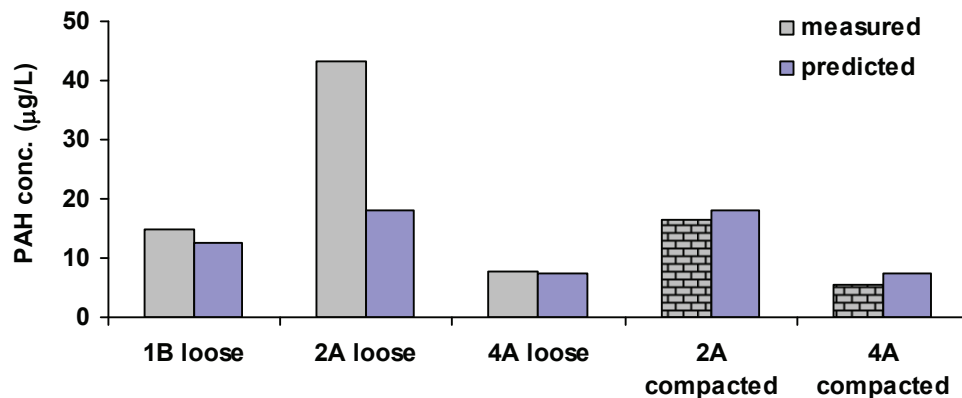
Although encapsulation and/or water exclusion (upon compaction) were initially considered to be important factors in immobilising PAHs, leaching results were more consistent with bitumen-PAH equilibrium partitioning being the major mechanism controlling PAH solubilisation in FB stabilised materials. In hindsight this is rather obvious, since PAHs are hydrophobic and exhibit strong binding to black carbon phases such as bitumen. An increase in the proportion of a strong binding phase (in this case 3% FB) will result in a lower proportion of PAHs partitioned into the aqueous phase in accordance with the following equation:

$$C_w = \frac{C_s}{K_{bitumen} \cdot f_{bitumen}}$$

– where concentration in water and solid are C_w and C_s , respectively; $K_{bitumen}$ = partitioning coefficient of PAH for bitumen; and $f_{bitumen}$ = fraction of bitumen in the sample.

Relative to C_w , C_s is so large it can be considered a constant, and therefore all other things being equal, the aqueous concentration of PAHs would be expected to be inversely proportional to the fraction of bitumen/binder present in the stabilised matrix. Using the above equation, $K_{bitumen}$ values from Brandt and de Groot (2001), and the assumption that non-FB samples contained ca 2.5% binder (40mm seal layer containing ca 8% binder) and FB samples (4A) contained 5.5% binder, the predicted total concentration for five major PAHs (fluorene, phenanthrene, anthracene, fluoranthene and pyrene) were calculated (figure 3.21). The partitioning equation does not distinguish between loose and compacted samples. In general, predicted concentrations were in good agreement with the actual leachate concentrations, although a noticeable exception was the loose ‘cement-only’ material (2A). The higher than predicted concentration in loose 2A is thought to be caused by the high DOC concentration (ca 20mg/L), which, as mentioned previously, can increase the solubilisation of PAHs by acting as a colloidal phase into which PAHs can partition. When compacted, the reduced surface area of 2A exposed to leaching resulted in decreased alkalinity (pH 9.7 vs 11.8) and lower DOC (ca 6mg/L). The lower DOC concentration presumably explains the closer fit between actual and predicted PAH concentrations for compacted sample 2A.

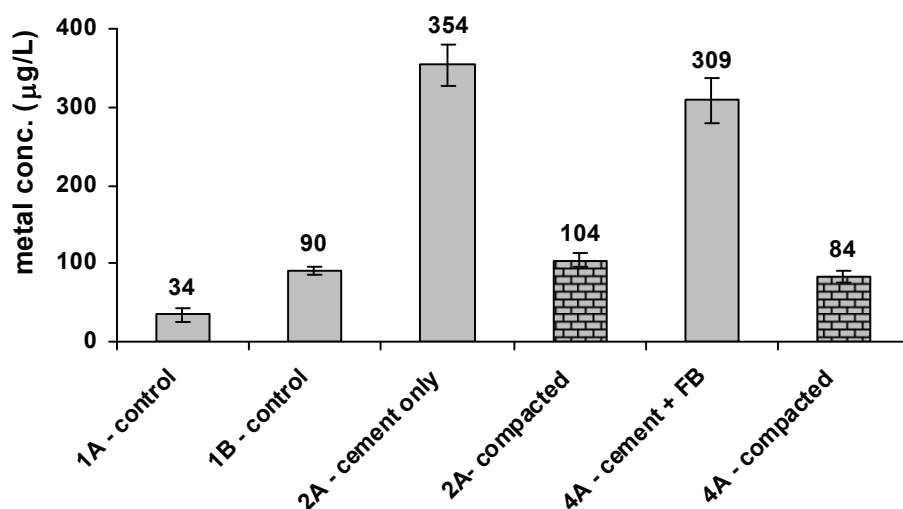
Figure 3.21 Predicted (partition theory) and actual leachate concentrations of five PAHs – fluorene, phenanthrene, anthracene, fluoranthene and pyrene (brick pattern denotes compacted material)



3.3.3.4 Leaching of heavy metals

The total concentration of heavy metals (chromium, copper, lead, nickel and zinc) in leachates ranged from $34\mu\text{g/L}$ – $354\mu\text{g/L}$ (refer to table A.8, appendix A for individual concentrations). Chromium and copper accounted for ca 90% (copper ca 15% and chromium ca 75%) of the total concentration of leachate heavy metals. The concentration of both these metals was highly correlated to leachate pH [$r^2 = 0.90$ (Cr); $r^2 = 0.93$ (Cu)]. For copper, whose solubility generally decreases with increasing pH, the correlation with pH most likely reflects a strong correlation between leachate pH and DOC ($r^2 = 0.97$) – with DOC mobilisation of copper via DOC-copper complexation (Impellitteri et al 2002; Jordan et al 1997; Temminghoff et al 1997; Yin et al 2002). The alkalinity of the leachate solutions kept metals like zinc at relatively low concentrations (4 – $14\mu\text{g/L}$).

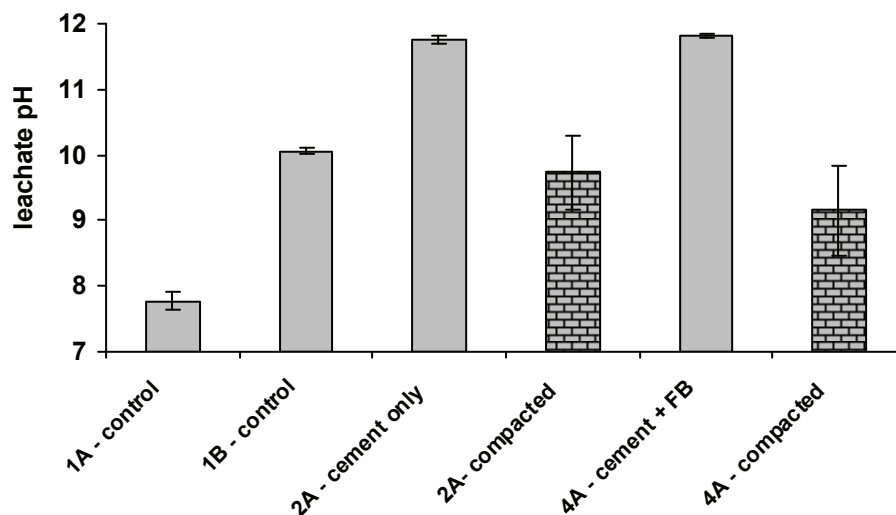
Figure 3.22 Leachate heavy metal concentrations from loose (plain grey bars) and compacted (brick patterned bars) stabilised materials



Unlike PAHs where partitioning to FB was the dominant mechanism controlling leaching, the presence of FB had no effect on the immobilisation of hydrophilic-dissolved metal ions either loose or compacted

samples. This was apparent from the similar leached concentrations of metals from 2A (cement only) and 4A (cement + FB) shown in figure 3.22. The observation that FB had no effect on reducing the leachate concentrations of metals suggests that the encapsulation effect in the loose material from the FB process is minimal. Compaction, on the other hand and irrespective of FB, resulted in ca four-fold decrease in the concentration of heavy metals in leachates. The reason for this was lower surface area resulting in less cement in contact with water, which in turn, resulted in leachate solutions with lower pH (figure 3.23) and lower DOC concentrations (ca 20 to 6–7mg/L). The lower pH and DOC concentrations in leachates from compacted samples accounted for the decreased solubilisation of chromium and copper, respectively. So in other words, compaction affects those processes (ie pH/DOC) that, in turn, affect PAH mobilisation.

Figure 3.23 pH of leachate solutions from aqueous extraction of loose and compacted Burke St roading materials. Brick pattern denotes compacted samples and error bars represent values from duplicate samples

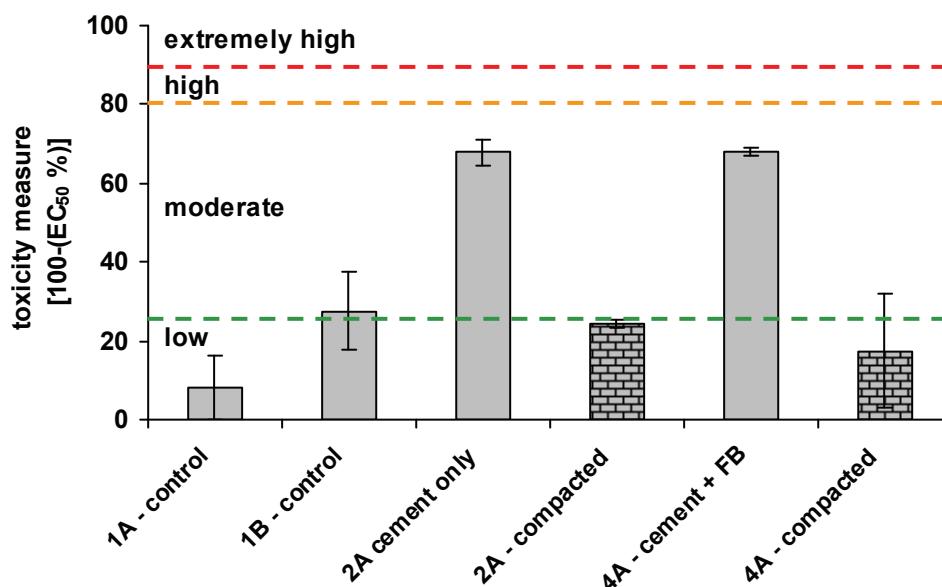


3.3.4 Toxicity testing of leachates

3.3.4.1 Effects concentrations (EC_{50}) for growth inhibition of *P. subcapitata*

The EC_{50} is the concentration (expressed as percentage of original leachate solution) that inhibited algal growth, relative to the control, by 50%. In other words, a 1/10 dilution of the original leachate causing 50% algal inhibition would correspond to an EC_{50} value of 10%. In an effort to present the toxicity data in a way that is consistent with previous figures showing contaminant concentrations (ie so that high values correspond to high toxicity and vice versa) the EC_{50} percent concentrations were subtracted from 100% - hence in the above example, the toxicity would be plotted as $100 - 10\% = 90\%$. The algal toxicities of the leachates, expressed as $[100 - EC_{50}\%]$ are summarised in figure 3.24 (3B is not shown since it was similar to 4A).

Figure 3.24 Mean (n=2) toxicity (EC₅₀ values) of leachates to the fresh water alga *P. subcapitata* (brick pattern denotes compacted samples); error bars = duplicate samples; dashed lines indicate the leachates 'potential for harm' (modified from Harrington-Hughes 2000)



The mean EC₅₀ values (% of original leachate concentration) ranged from the 92% for 1A (control) to the most toxic value of 32% for loose material of 2A and 4A. As mentioned, as a control, sample 1B was considered to be more representative than 1A of non-stabilised material, and had a leachate EC₅₀ value of 72%. Given the low L/S ratio of 2, the toxicity of the leachates was relatively low given that the most toxic EC₅₀ corresponded to only a three-fold dilution of the original leachate concentration. In contrast, using the same testing method with leachates from road sweepings and using a much more diluted L/S ratio of 20, Depree (2008) reported LC₅₀ values as low as 2.2% (or 45-fold dilution) of the original leachate.

Harrington-Hughes (2000) conducted leaching tests on numerous materials used in road construction and/or repairs to evaluate potential risks to receiving waters. The study included materials such as asphalt cement, Portland cement, aggregate, fly ash, slag and scrap tyres, with an L/S ratio of 4 used for the leaching experiments. Toxicity tests were conducted using *Daphnia* and the same freshwater algae, *P. subcapitata*, and based on the results, a relationship between the EC₅₀ of leachates and 'potential for harm' was proposed (table 3.3). Applying these definitions to the Burke St samples (figure 3.24), the potential for harm from the leachates ranged from low to moderate.

Table 3.3 Relationship between 'Potential for harm' and leachate EC₅₀ values (Harrington-Hughes 2000)

Potential for harm	EC ₅₀ value (%)	[100-EC ₅₀] value (%)
Extremely high	<10	>90
High	10-20	80-90
Moderate	20-75	25-80
Low	>75	<25

It is interesting to note that leachates of unstabilised materials (controls 1A and 1B) had a low potential for harm, and that the addition of stabilisers actually increased the potential risk of the leachates (towards the upper limit of 'moderate'). This does not imply that stabilisation is not beneficial, as without stabilisation,

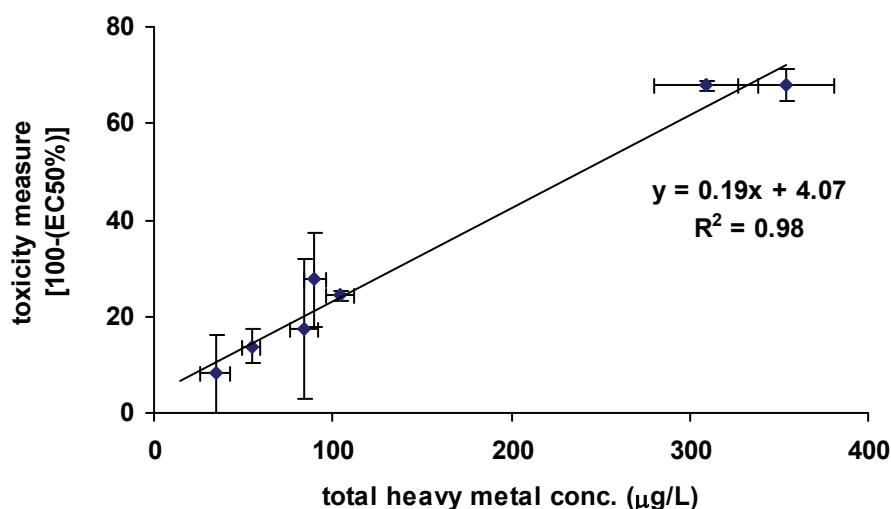
it would not be possible to reuse the tar-contaminated material as a semi-bound base for the new road. However, it does indicate that removal and temporary storage of tar-contaminated material does not represent a major risk with respect to contaminant leaching – although obviously measures should still be in place to minimise the loss of any particulate material. This could be beneficial in circumstances where in-situ stabilisation is not suitable, for example, weak subgrade and/or a road with unsuitable crossfall (transverse slope of the roadway), which prevents the addition of make-up aggregate.

While the toxicity results indicate there is a low risk of harm from contaminant leaching, this does not imply the solid material is not potentially harmful. As discussed in section 1.3.4, tar-contaminated roading materials have been shown to be toxic to soil-dwelling insects (LC₅₀ value 300mg/kg; Depree 2007), and mutagenicity index (MI) values of 1 and 2, corresponding to PAH concentrations of 140 and 370mg/kg, respectively, have been reported by Blackburn et al (1999). The benefit of reusing the material as a heavily compacted, semi-bound road base material is that the material is no longer ecologically relevant, that is, it is bound into a densely compacted matrix sealed beneath a bitumen wearing course. As such, the particulate toxicity in this application becomes largely irrelevant. An additional benefit of an intended reuse application involving a compacted material is that the highest leachate toxicity from cement-containing materials (2A and 4A) was reduced two to three fold (to control levels) when compacted. The reuse of tar-contaminated material in compacted road base therefore addresses environmental concerns regarding the toxicity of leachates and the particulate material itself.

3.3.4.2 Source of 'moderate' leachate toxicity

The similar plots of leachate toxicity (figure 3.24) and heavy metal concentrations (figure 3.23) strongly suggest that leachate toxicity is attributable to heavy metals rather than the hydrophobic PAHs associated with coal tar binders. This hypothesis is supported by a strong correlation between toxicity and heavy metals ($R^2 = 0.98$; figure 3.25), whereas PAHs were only weakly correlated to toxicity ($R^2 = 0.33$, data not shown). Note that leachate pH can be discounted as a source of toxicity since the pH of all leachates was adjusted to 6.8–7.0 prior to carrying out the toxicity assays.

Figure 3.25 Correlation between total concentration of heavy metals and toxicity (error bars represent values from duplicate samples)



PAHs

There are very few toxicity studies involving dissolved PAHs, mainly because they are hydrophobic contaminants with low aqueous solubilities and hence mainly associated with sediment/particulate phases. Šepič et al (2003) reported an algal growth inhibition EC_{50} for fluoranthene of $192\mu\text{g/L}$, which was ca 30 times greater than the maximum concentration of $6.7\mu\text{g/L}$ (2A) measured in the Burke St leachates. The acute toxicity of acenaphthene has been investigated using the dipternha larva, *Paratanytarus parthenogeneticus* with no lethal acute effect observed in saturated ($2100\mu\text{g/L}$) solutions of acenaphthene (Meiser et al 2000). Of the 16 United States EPA-listed PAHs, Schirmer et al (1998) found that only two- and three-ringed PAHs were directly cytotoxic to a cell line from the rainbow trout gill. Of the cytotoxic PAHs (naphthalene, acenaphthylene, acenaphthene, fluorene and phenanthrene), only naphthalene was found to be effective (toxic) at concentrations well below its water solubility. Based on the absence of acute PAH toxicity in previous studies, and on the finding that the highest total leachate concentration of PAHs was only $52\mu\text{g/L}$ for tar-contaminated Burke St materials, it seems unlikely that the moderate leachate toxicity was attributable to tar-derived PAHs.

Eom et al (2007) undertook a range of toxicity studies on leachates from a contaminated cokery soil, containing 2634 mg/kg of PAHs (16 US EPA), that yielded a leachate concentration of ca $18\mu\text{g/L}$. Despite the high concentration of PAHs in the particulate phase, the authors concluded that the observed toxicity on aquatic organisms was attributable to inorganic water-extractable pollutants, in particular copper. However, soil toxicity towards earthworms and collembolae (ie *F. candida*) could be explained by the PAHs, which is consistent with the reported LC_{50} value of ca 300mg/kg for *F. candida* using tar-contaminated soils (Depre 2007).

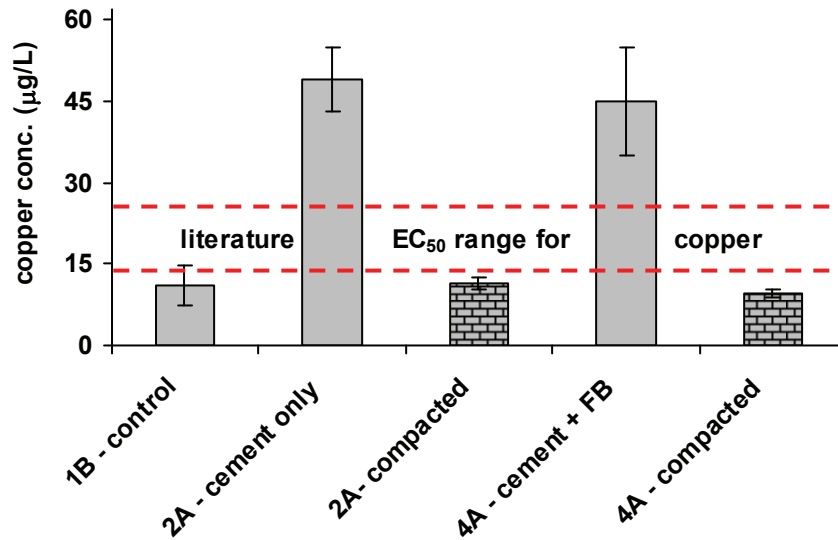
Phenols

Phenols are another contaminant class that is present in relatively high concentrations in coal tar-contaminated materials. Although not measured in this study, it was considered unlikely that phenolic compounds would be present at sufficient concentrations to result in toxicity given that the growth inhibition (using *P. subcapitata*) EC_{50} for phenol is reportedly $20,900\mu\text{g/L}$ (Chen and Lin 2006).

Heavy metals

Leachate toxicity was strongly correlated to the concentration of total heavy metals (figure 3.25). The two most abundant heavy metals, which accounted for ca 90% of the total and therefore were the most likely to be present at toxic concentrations, were chromium and copper. Leachate concentrations of chromium from stabilised compacted and loose samples were ca 70 and $260\mu\text{g/L}$, respectively. For copper, the respective leachate concentrations from compacted and loose samples were ca 11 and $45\mu\text{g/L}$. Leachate toxicity was strongly correlated to both chromium ($R^2 = 0.98$) and copper ($R^2 = 0.99$) concentrations; however, copper was considered the most likely cause of elevated toxicity because of its inherent toxicity at low concentrations to aquatic organisms. For example, De Schampelaere et al (2004) reported *P. subcapitata* growth inhibition EC_{50} values of $15\text{--}25\mu\text{g/L}$ for copper (pH 7–8), which is consistent with the aquatic organism HC_{50} value of $18\mu\text{g/L}$ (Lijzen et al 2001). De Schampelaere et al (2004) also reported a *D. magna* EC_{50} of $21.6\mu\text{g/L}$ (pH 7) for copper. As shown in figure 3.26, the leachate concentrations of copper in loose Burke St samples 2A and 4A are consistent with the observed toxicity. The copper concentrations and EC_{50} leachate % values corresponded to copper EC_{50} values for 2A and 4A (loose material) of 17.7 and $14.4\mu\text{g/L}$, respectively, which is comparable to the range reported by De Schampelaere (2004). As mentioned, Eom (2007) concluded that aquatic toxicity of leachates from a heavily PAH-contaminated cokery (ie tar) soil was attributable to solubilised copper and not PAHs.

Figure 3.26 Mean leachate concentrations of copper from Burke St samples. Dashed lines indicate reported copper EC₅₀ range for *P. subcapitata* (De Schamphelaere et al 2004). Error bars represent values from duplicate samples



Chromium (III) is less toxic than copper; for example, Michnowicz and Weaks (1984) and Labra et al (2007) reported *P. subcapitata* growth inhibition EC₅₀ values for chromium (III) of ca 600µg/L and ca 870µg/L, respectively. As such, even the highest leachate concentrations of chromium (III) from Burke St samples (ca 260µg/L) were significantly lower than published EC₅₀ values. This suggests that chromium is less likely to be the source of leachate toxicity than copper.

Although this section has detailed some low-to-moderate risk of leachates from stabilised tar-containing materials, it is important to emphasise the following points:

- The leachates were very concentrated (L/S of 2) and toxicity was generally low.
- Leachate toxicity was only 'moderate' for non-compacted (loose) stabilised materials. In the intended application the materials would always be compacted, and leachate toxicity for compacted samples corresponded to low toxicity.
- Leachate toxicity was not attributable to the coal tar-derived PAHs.
- Toxicity was correlated with heavy metals, and in particular copper the origin of which we believe is independent of coal tar contamination. This is because metals are not commonly associated with coal tar (as opposed to gas works waste) as it is a distillation product from coal carbonisation.
- Given that the application is compacted road base material, an environmentally irrelevant compartment, the only environmental concern is the potential impact of leaching/runoff water on receiving waters (ground or surface). In most cases this is very unlikely because the compacted road base is overlain with a water-impervious seal layer (ie asphalt or chip seal).

3.3.5 Practical aspects of the Burke St stabilisation

The previous sections (3.3.2 to 3.3.4) have shown that, with respect to environmental concerns/risks of reconstruction of tar-contaminated roads, the concept of in-situ recycling with cement + FB stabilisation has considerable merit. However, in order to be a feasible solution, it is necessary for the method to provide a reconstructed road base with the required strength and properties to achieve specified design lifetimes. Otherwise, if the road fails prematurely, this can potentially create larger problems. For example, if the road failed and the entire recycled road base required excavation and landfill, it would require disposal in landfill of ca 150mm depth of tar-contaminated material, as opposed to the original 40mm of seal and 20–30mm of underlying contaminated basecourse. Even if road failure just resulted in additional maintenance or use of more expensive surface treatments (eg asphalt rather than chip seal), all of these factors would add to the life-cycle cost analysis (ie whole-life cost), which might affect the long-term economic feasibility of the method.

From the outset of the project, one of the main challenges for implementing in-situ recycling on old tar-contaminated roads was the requirement to have an approximately 150mm thick layer of construction thickness (seal + basecourse). In the absence of sufficient construction thickness, it is necessary to apply make-up aggregate in order to attain the necessary strength and thickness in the rehabilitated pavement without incorporating underlying subgrade material. Incorporation of excessive amounts of subgrade clay/silt results in a ‘dirty’ surface, which creates adhesion problems for the overlying chip seal wearing course. While cross-fall can limit the amount of make-up aggregate that can be used, in the case of Burke St, there was no limitation, and given the marginal thickness of the construction layer, in hindsight, it would have been prudent to add 50mm of make-up aggregate. Ultimately, however, this decision was the responsibility of the contractor (Fulton Hogan), and based on preliminary investigations, they concluded (based on the information provided to them from City Care) that Burke St contained sufficient construction thickness to proceed without the addition of make-up aggregate.

Figure 3.27 Recycled pavement layer of Burke St showing a high amount of fine clay in the surface (a). A high proportion of clay is clearly evident from the largely ‘dirt-like’ appearance of the surface of a compacted sample of 4A (b)





From the photographs taken during the Burke St pavement rehabilitation (figure 3.27a), the recycled pavement appeared to contain a high proportion of silt/clay material, presumably because of incorporated subgrade material. The 'dirt-like' surface of the compacted recycled material is shown more clearly in figure 3.27b.

Surfaces with a high proportion of clay/silt have a dust layer between the road base aggregate and the bitumen binder of the overlying chip seal sealing, which prevents adhesion between the two layers. To address these concerns, Fulton Hogan contractors rolled stone chip into the surface prior to applying the bitumen chip seal layer. Approximately nine months after the Burke St in-situ stabilisation trial, Christchurch City Council personnel reported that the road was suffering from chip seal delamination (figure 3.28). It was noted that the seal was not able to cope with power-steering stresses or street-sweeper cleaning. With regards to the stabilised road base layer, it was noted that it had reasonable strength (although weaker than anticipated), but there was evidence of reduced strength where the seal had delaminated.

Figure 3.28 Photos (S.McNeill, Christchurch City Council) taken January 2009 (ca nine months after the in-situ pavement recycling) showing chip seal delamination – indicative of weak adhesion between the 'silty' FB-stabilised base and the overlying bituminous chip seal wearing course



Clearly the overall performance of the rehabilitated pavement was not ideal, especially since the whole approach was to lock up the tar-contaminated material in the carriageway beneath a water-impervious seal layer. The reality, however, was that in less than nine months patches of tar-contaminated basecourse material were exposed, thereby exacerbating the potential for erosion (particulate losses), leaching, and accelerated damage of the FB-stabilised base from water infiltration. Assuming Burke St is representative of tar-contaminated roads in Christchurch (and potentially other parts of New Zealand), then unless make-up aggregate can be applied to increase construction thickness – or an alternative method can be found that addresses the problems encountered in the Burke St trial – in-situ recycling may not be a feasible solution for reconstructing tar-contaminated streets. To realise the potential cost benefits (environmental and economic) of reusing tar-contaminated road materials, it may therefore be necessary to investigate alternative methods. Although not as desirable, the low risk of leaching from stored tar-contaminated material may enable methods where the material can be removed and reapplied after reshaping and/or modification of subgrade material.

4 Summary and conclusions

The aims of the research were two-fold, first to analyse the extent of the issues relating to tar contaminated roads, and second, to trial and test in-situ FB/cement stabilisation techniques to immobilise potentially toxic tar-derived PAHs. The hypothesis was that the inclusion of bitumen would reduce the leaching of tar-associated PAHs by encapsulation and/or exclusion of water when in a compacted matrix (as the material would be in a recycled road base application).

Conclusions about the effectiveness of FB stabilisation to immobilise PAHs were complicated by the practical requirement of ca 1.5% cement in stabilisation mixes. The cement increased leachate pH, resulting in higher concentrations of DOC, which, in turn, increased the mobilisation of PAHs and copper. Accordingly, to assess the effectiveness of FB stabilisation to reduce mobility of tar-derived PAHs, it was more relevant to compare the results to those of the 'cement only' (2A) control, as opposed to the unstabilised control (1B) so as to have comparable leachate conditions.

Relative to the 'cement-only' stabilised sample (2A), the inclusion of 3% FB (4A) resulted in a ca five-fold reduction in the leachate concentration of PAHs in loose (uncompacted) material. Compaction of the sample, which was initially hypothesised as the principle mechanism for reducing contaminant mobility, had only a minor effect on immobilising PAHs in FB-stabilised samples. However, the decreased mobilisation of PAHs in FB-stabilised samples was consistent with, and predicted by, equilibrium partition theory, and this process was more likely to be the cause of the results observed than was encapsulation or matrix impermeability to water. In other words, increasing the fraction of bituminous carbon resulted in increased binding/partitioning of PAHs to the particulate phase, and, consequently, decreased dissolved concentrations in the aqueous phase.

Compaction was more effective in reducing (three-fold) the solubilisation of PAHs from the cement only sample (2A) because exclusion of water resulted in a lower leachate pH, which, in turn, yielded less DOC for enhancing the mobilisation of PAHs.

Aqueous leachates of stabilised materials exhibited only low-to-moderate leachate toxicity to the freshwater algae *P.subcapitata* (note: alkaline pH neutralised prior to testing). Leachate toxicity was greatest for the loose samples, irrespective of whether they contained FB (4A) or not (2A), and decreased ca three-fold when the material was compacted. The incorporation of FB had a minimal 'encapsulation effect', based on similar metal solubilisation and leachate toxicity of samples with FB (4A) and without FB (2A). Had there been effective encapsulation, presumably there would have been a decrease in the surface area of the metal-containing particulates exposed to the leaching solution. The leachate toxicity of both types of materials ('cement only' and 'cement + FB') was classified as low when compacted, which is the final state of material in the intended application. Leachate toxicity was not attributable to tar-derived PAHs, but rather to copper present at concentrations of ca 45 and 11 µg/L for compacted and loose samples, respectively. The observed copper EC₅₀ values of 14–18 µg/L were consistent with reported toxicity values. Although beyond the scope of the research programme to confirm, it is assumed that the aggregate and/or subgrade minerals are the source of heavy metals; however, additional work is need to confirm this.

Based on contaminant leaching and toxicity, the reuse of tar-contaminated roads as compacted stabilised base material represents minimal risk to the environment. In addition to very low potential for harm from the leachates of reused tar-contaminated material, reuse of the contaminated material as road base limits

exposure because a) it is capped with a water-proof seal layer and b) it is beneath a carriageway and thus is not an environmentally significant compartment.

The major drawback of the method was that the thickness of available construction layers and/or quality of the basecourse aggregate was probably not sufficient for in-situ recycling methods. The Burke St trial resulted in a compacted road base with a very 'silty' surface that prevented good adhesion with the overlying bitumen chip seal layer. Consequently, within nine months, some delamination of the seal had occurred, which exposed the tar-containing stabilised road base. This is clearly undesirable for reasons of isolating the material within the carriageway, and therefore a modified approach (such as the application of make-up aggregate) is required to address this limitation. If the cross fall is such that it is not possible to add additional aggregate, then alternative reuse options involving excavation and off-site processing/stabilisation would need to be considered. However, the feasibility of such methods, and how they compare with landfill disposal, was beyond the scope of the research project.

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Appendix A: Analytical data tables

Table A.1 PAH concentrations (mg/kg) in road materials from Newcastle, Ryan, Glasgow and Burke St (four sites samples along selected length of each street)

PAH	Newcastle 1	Newcastle 2	Newcastle 3	Newcastle 4
Naphthalene	13.7	4.4	0.4	1.2
Acenaphthene	37.9	14.0	0.5	3.2
Acenaphthylene	9.9	3.6	0.6	1.9
Fluorene	75.3	26.2	0.6	5.0
Anthracene	134.3	53.8	1.8	10.0
Phenanthrene	634.6	132.1	9.9	51.4
Fluoranthene	777.5	181.4	36.9	93.6
Pyrene	739.0	166.4	29.4	82.0
Benzo[a]anthracene	258.8	115.7	11.3	32.7
Chrysene	324.2	92.5	11.2	25.6
Benzo[b]fluoranthene	442.3	140.9	23.8	48.9
Benzo[k]fluoranthene	153.3	50.6	7.2	15.9
Benzo[a]pyrene (BAP)	263.7	126.7	12.6	31.2
Indenopyrene	122.0	67.0	9.1	20.5
Dibenzo[a,h]anthracene	35.7	18.6	2.5	5.8
Benzo[g,h,i]perylene	166.5	95.6	11.9	25.1
Total	4188.7	1289.4	169.7	454.1
PAH	Ryan 1	Ryan 2	Ryan 3	Ryan 4
Naphthalene	6.5	16.7	6.8	2.3
Acenaphthene	19.6	13.2	6.5	2.0
Acenaphthylene	7.5	4.8	11.9	3.0
Fluorene	33.3	30.9	28.2	6.1
Anthracene	69.9	50.2	38.4	8.9
Phenanthrene	330.1	265.6	174.1	46.5
Fluoranthene	493.5	256.0	177.6	80.3
Pyrene	467.3	236.8	171.8	72.2
Benzo[a]anthracene	145.4	77.8	65.3	32.6
Chrysene	165.7	90.9	70.1	23.5
Benzo[b]fluoranthene	220.6	130.6	102.0	41.5
Benzo[k]fluoranthene	86.3	50.0	40.5	14.3
Benzo[a]pyrene (BAP)	162.7	89.7	65.3	26.3
Indenopyrene	74.2	42.3	32.7	15.6
Dibenzo[a,h]anthracene	20.3	11.7	8.8	4.7
Benzo[g,h,i]perylene	103.3	61.5	40.1	21.4
Total	2406.2	1428.7	1040.1	401.3

Table A.1 (cont) PAH concentrations (mg/kg) in road materials from Newcastle, Ryan, Glasgow and Burke St (four sites samples along selected length of each street)

PAH	Burke 1	Burke 2	Burke 3	Burke 4
Naphthalene	4.9	8.7	4.0	11.9
Acenaphthene	16.9	26.9	16.1	31.1
Acenaphthylene	4.2	6.5	4.4	7.7
Fluorene	24.0	37.7	18.1	44.9
Anthracene	47.5	91.1	43.5	110.6
Phenanthrene	218.9	410.0	229.8	561.4
Fluoranthene	323.5	572.7	398.0	689.4
Pyrene	306.4	485.9	356.0	619.5
Benzo[a]anthracene	108.1	179.2	135.5	228.7
Chrysene	121.6	182.0	140.7	245.7
Benzo[b]fluoranthene	196.3	290.7	227.0	413.0
Benzo[k]fluoranthene	71.1	99.3	83.9	129.4
Benzo[a]pyrene (BAP)	120.8	173.5	132.7	215.0
Indenopyrene	62.5	84.8	64.9	104.3
Dibenzo[a,h]anthracene	17.4	19.3	15.7	25.3
Benzo[g,h,i]perylene	84.3	122.3	96.0	155.3
Total	1728.4	2790.7	1966.5	3593.0
PAH	Glasgow 1	Glasgow 2	Glasgow 3	Glasgow 4
Naphthalene	2.3	1.0	3.5	7.9
Acenaphthene	4.6	6.0	8.7	13.8
Acenaphthylene	7.1	4.4	3.6	7.1
Fluorene	10.6	11.3	9.8	25.1
Anthracene	17.1	18.9	22.2	72.8
Phenanthrene	78.3	90.2	95.3	309.5
Fluoranthene	135.0	168.8	168.2	489.4
Pyrene	120.7	149.2	172.3	489.4
Benzo[a]anthracene	64.0	66.8	83.3	181.7
Chrysene	46.3	50.8	67.9	200.0
Benzo[b]fluoranthene	99.0	76.3	121.4	314.8
Benzo[k]fluoranthene	32.9	27.3	40.3	110.8
Benzo[a]pyrene (BAP)	77.3	51.9	94.3	191.0
Indenopyrene	40.7	25.6	45.6	88.6
Dibenzo[a,h]anthracene	13.7	8.5	14.5	24.9
Benzo[g,h,i]perylene	55.0	34.6	67.9	132.8
Total	804.5	791.5	1018.8	2659.8

Table A.2 PAH concentrations (mg/kg) in Burke St design mixes – prepared by Fulton Hogan Waikato laboratory (note all FB-stabilised samples also contained 1.5% cement)

PAH	1% FB	2%FB	4% FB	Control
Naphthalene	3.1	3.4	3.6	3.8
Acenaphthene	9.2	8.0	8.2	10.4
Acenaphthylene	8.0	9.4	10.1	11.2
Fluorene	11.9	10.4	10.5	13.4
Anthracene	33.5	31.9	30.2	39.9
Phenanthrene	130.0	111.6	117.4	143.7
Fluoranthene	205.5	199.2	213.5	239.5
Pyrene	230.6	219.1	231.3	259.5
Benzo[a]anthracene	75.5	77.7	83.6	93.8
Chrysene	75.5	77.7	83.6	91.8
Benzo[b]fluoranthene	142.6	153.4	165.5	175.6
Benzo[k]fluoranthene	50.3	59.8	65.8	73.9
Benzo[a]pyrene (BAP)	77.6	77.7	83.6	93.8
Indenopyrene	48.2	49.8	53.4	59.9
Dibenzo[a,h]anthracene	20.1	21.9	23.1	22.0
Benzo[g,h,i]perylene	52.4	61.8	65.8	71.9
Total	1174	1173	1249	1404

Table A.3 Leachate PAH concentrations (µg/L) in design mix samples from Burke St (note all FB-stabilised samples also contained 1.5% cement)

PAH	1% FB	2%FB	4% FB	Control
Naphthalene	1.4	1.2	0.8	<0.5
Acenaphthene	3.8	3.1	1.7	3.6
Acenaphthylene	0.5	0.5	0.4	0.4
Fluorene	3.3	2.5	1.4	1.1
Anthracene	2.7	3.8	2.2	1.5
Phenanthrene	12.0	10.0	5.1	<0.4
Fluoranthene	4.7	4.1	2.0	5.6
Pyrene	2.9	3.4	1.6	4.3
Benzo[a]anthracene	0.2	0.3	0.2	0.4
Chrysene	0.2	0.2	<0.1	0.3
Benzo[b]fluoranthene	<0.1	<0.1	<0.1	0.4
Benzo[k]fluoranthene	<0.1	<0.1	<0.1	0.1
Benzo[a]pyrene (BAP)	<0.1	<0.1	<0.1	0.2
Indenopyrene	<0.1	<0.1	<0.1	0.2
Dibenzo[a,h]anthracene	<0.1	<0.1	<0.1	<0.1
Benzo[g,h,i]perylene	<0.1	<0.1	<0.1	0.2
Total	31.6	29.1	15.3	18.3

Table A.4 Leachate heavy metal concentrations ($\mu\text{g/L}$) in design mix samples from Burke St (note all FB-stabilised samples also contained 1.5% cement)

Heavy metal	1% FB	2%FB	4% FB	Control
arsenic	3.3	4.1	3.6	<1.1
cadmium	0.065	<0.053	<0.053	0.053
chromium	77	71	68	<0.53
copper	77	68	80	4.2
lead	<0.11	<0.11	<0.11	0.38
nickel	22	20	23	<5.3
zinc	<11	6.3	8.6	21

Table A.5 PAH concentrations (mg/kg) in field-stabilised samples from Burke St trial. Replicate samples represented as i, ii and iii

PAH	1A (i)	1A (ii)	1A (iii)	1B (i)	1B (ii)
Naphthalene	1.8	1.4	1.9	2.3	1.6
Acenaphthene	3.4	2.9	4.7	6.7	3.6
Acenaphthylene	3.1	3.0	3.1	4.9	3.5
Fluorene	4.5	3.9	6.6	9.9	5.1
Anthracene	14.2	12.3	87.1	26.2	14.6
Phenanthrene	59.9	49.3	18.6	117.7	61.3
Fluoranthene	93.5	84.1	125.7	159.9	102.1
Pyrene	92.0	84.1	117.1	159.9	99.2
Benzo[a]anthracene	30.7	29.0	42.9	58.1	33.6
Chrysene	36.5	33.3	44.3	61.0	39.4
Benzo[b]fluoranthene	68.6	65.2	50.0	110.4	75.9
Benzo[k]fluoranthene	27.7	26.1	28.6	34.9	32.1
Benzo[a]pyrene (BAP)	45.3	43.5	81.4	75.6	51.1
Indenopyrene	30.7	29.0	31.4	42.1	33.6
Dibenzo[a,h]anthracene	14.6	11.9	12.3	16.0	14.6
Benzo[g,h,i]perylene	27.7	27.5	37.1	43.6	30.6
Total	554.0	506.7	692.9	929.2	601.7

Table A.5 (cont) PAH concentrations (mg/kg) in field-stabilised samples from Burke St trial. Replicate samples represented as i, ii and iii

PAH	1B (iii)	2A (i)	2A (ii)	2A (iii)	3B (i)
Naphthalene	1.7	3.5	3.0	2.1	3.1
Acenaphthene	4.8	7.7	6.3	7.9	7.4
Acenaphthylene	3.8	5.5	4.9	4.7	6.3
Fluorene	5.9	11.1	9.4	12.6	10.6
Anthracene	79.7	29.5	25.1	147.1	26.5
Phenanthrene	17.4	144.5	115.2	30.9	120.6
Fluoranthene	127.5	191.7	147.6	176.5	176.6
Pyrene	118.8	176.9	144.7	161.8	176.6
Benzo[a]anthracene	43.5	66.4	53.2	60.3	63.3
Chrysene	46.4	70.8	57.6	61.8	66.2
Benzo[b]fluoranthene	52.2	128.3	104.8	67.6	119.2
Benzo[k]fluoranthene	30.4	41.3	34.0	36.8	39.7
Benzo[a]pyrene (BAP)	88.4	85.5	70.9	111.8	89.7
Indenopyrene	34.8	51.6	45.8	44.1	47.1
Dibenzo[a,h]anthracene	13.8	22.1	19.2	17.6	16.2
Benzo[g,h,i]perylene	37.7	50.1	45.8	48.5	51.5
Total	706.8	1086.4	887.4	992.1	1020.5
PAH	3B (ii)	3B (iii)	4A (i)	4A (ii)	4A (iii)
Naphthalene	2.3	1.9	2.8	2.5	3.1
Acenaphthene	4.8	4.9	7.3	6.0	8.2
Acenaphthylene	5.0	4.1	5.4	4.8	5.7
Fluorene	6.9	6.2	10.4	8.3	10.9
Anthracene	20.5	80.9	26.4	21.9	139.7
Phenanthrene	77.7	17.6	126.1	93.5	27.9
Fluoranthene	127.6	126.5	161.2	127.0	176.5
Pyrene	124.6	117.6	161.2	124.1	176.5
Benzo[a]anthracene	46.9	45.6	57.2	46.7	64.7
Chrysene	49.8	47.1	61.6	48.2	64.7
Benzo[b]fluoranthene	89.4	52.9	104.1	84.7	69.1
Benzo[k]fluoranthene	30.8	32.4	35.2	29.2	39.7
Benzo[a]pyrene (BAP)	67.4	98.5	71.8	61.3	123.5
Indenopyrene	30.8	35.3	26.4	23.4	45.6
Dibenzo[a,h]anthracene	12.9	14.3	11.6	10.7	19.1
Benzo[g,h,i]perylene	36.7	39.7	42.5	36.5	48.5
Total	734.3	725.4	911.1	728.8	1023.5

Table A.6 PAH concentrations ($\mu\text{g/L}$) in leachates of *non-compacted* field-stabilised samples from Burke St trial

PAH	1A (i)	1A (ii)	1B (i)	1B (ii)	2A (i)
Naphthalene	0.00	0.00	0.64	0.62	3.40
Acenaphthene	0.04	0.03	2.60	2.60	6.00
Acenaphthylene	0.08	0.07	0.26	0.23	0.50
Fluorene	0.00	0.03	2.40	1.60	6.40
Anthracene	0.02	0.21	3.90	0.06	22.00
Phenanthrene	0.30	0.02	2.20	1.80	4.90
Fluoranthene	0.02	0.03	4.30	4.30	7.50
Pyrene	0.03	0.02	4.90	3.90	8.40
Benzo[a]anthracene	0.02	0.02	0.37	0.30	0.57
Chrysene	0.01	0.02	0.24	0.24	0.35
Benzo[b]fluoranthene	0.10	0.09	0.13	0.18	0.15
Benzo[k]fluoranthene	0.02	0.02	0.04	0.07	0.04
Benzo[a]pyrene (BAP)	0.07	0.05	0.09	0.12	0.07
Indenopyrene	0.06	0.06	0.03	0.06	0.00
Dibenzo[a,h]anthracene	0.03	0.02	0.02	0.02	0.00
Benzo[g,h,i]perylene	0.07	0.09	0.03	0.06	0.00
Total	0.9	0.8	22.1	16.2	60.3
PAH	2A (ii)	3B (i)	3B (ii)	4A (i)	4A (ii)
Naphthalene	1.70	0.77	0.67	1.10	1.20
Acenaphthene	4.10	1.50	1.30	1.10	1.10
Acenaphthylene	0.50	0.20	0.17	0.13	0.14
Fluorene	4.90	1.50	1.20	1.10	1.00
Anthracene	18.00	5.00	4.00	3.50	3.50
Phenanthrene	3.60	1.40	1.20	0.93	1.00
Fluoranthene	5.90	1.80	1.80	0.99	1.30
Pyrene	4.60	2.10	1.40	1.20	0.97
Benzo[a]anthracene	0.28	0.16	0.15	0.11	0.11
Chrysene	0.19	0.09	0.09	0.06	0.07
Benzo[b]fluoranthene	0.08	0.04	0.06	0.03	0.06
Benzo[k]fluoranthene	0.02	0.01	0.01	0.01	0.02
Benzo[a]pyrene (BAP)	0.04	0.03	0.03	0.02	0.03
Indenopyrene	0.01	0.00	0.01	0.00	0.00
Dibenzo[a,h]anthracene	0.01	0.00	0.00	0.00	0.00
Benzo[g,h,i]perylene	0.01	0.00	0.01	0.00	0.02
Total	43.9	14.6	12.1	10.3	10.5

Table A.7 PAH concentrations ($\mu\text{g/L}$) in leachates of *compacted* field-stabilised samples from Burke St trial. Duplicate samples represented as 'i' and 'ii'

PAH	2A (i)	2A (ii)	3B (i)	3B (ii)
Naphthalene	0.61	0.40	0.25	0.15
Acenaphthene	1.20	1.00	0.76	0.59
Acenaphthylene	0.11	0.10	0.07	0.06
Fluorene	1.40	1.30	0.86	0.66
Anthracene	1.50	1.50	0.87	0.60
Phenanthrene	6.60	7.40	3.80	3.00
Fluoranthene	3.20	3.60	1.60	1.20
Pyrene	3.60	3.00	1.90	0.97
Benzo[a]anthracene	0.32	0.25	0.18	0.10
Chrysene	0.22	0.21	0.11	0.09
Benzo[b]fluoranthene	0.09	0.07	0.08	0.07
Benzo[k]fluoranthene	0.03	0.02	0.02	0.02
Benzo[a]pyrene (BAP)	0.05	0.03	0.06	0.03
Indenopyrene	0.00	0.00	0.02	0.02
Dibenzo[a,h]anthracene	0.00	0.00	0.00	0.01
Benzo[g,h,i]perylene	0.00	0.00	0.02	0.02
Total	18.9	18.9	10.6	7.6
PAH	4A (i)	4A (ii)		
Naphthalene	0.28	0.08		
Acenaphthene	0.55	0.25		
Acenaphthylene	0.05	0.03		
Fluorene	0.67	0.35		
Anthracene	0.63	0.49		
Phenanthrene	2.90	2.00		
Fluoranthene	1.10	0.96		
Pyrene	1.20	0.72		
Benzo[a]anthracene	0.12	0.07		
Chrysene	0.07	0.07		
Benzo[b]fluoranthene	0.04	0.04		
Benzo[k]fluoranthene	0.02	0.01		
Benzo[a]pyrene (BAP)	0.02	0.01		
Indenopyrene	0.00	0.00		
Dibenzo[a,h]anthracene	0.00	0.00		
Benzo[g,h,i]perylene	0.00	0.01		
Total	7.6	5.1		

Table A.8 Metal concentrations ($\mu\text{g/L}$) in leachates of field-stabilised samples from Burke St trial. Duplicate samples represented as 'i' and 'ii'

Heavy metal	1A(i)	1A(ii)	1B (i)	1B (ii)
arsenic	0.0	0.0	6.9	6.6
cadmium	0.0	0.0	0.0	0.0
chromium	0.7	0.0	64.0	69.0
copper	12.0	2.6	9.3	13.0
lead	0.8	0.2	0.0	0.3
nickel	0.0	4.1	0.0	1.5
zinc	29.0	19.0	4.3	5.4
Heavy metal	2A (i) non-	2A (ii) non-	3B (i) non-	3B (ii) non-
arsenic	6.3	5.6	7.0	6.9
cadmium	0.1	0.0	0.1	0.0
chromium	250.0	300.0	150.0	170.0
copper	46.0	52.0	35.0	47.0
lead	0.0	0.0	0.1	1.1
nickel	11.0	16.0	0.9	5.7
zinc	14.0	7.2	4.5	6.8
Heavy metal	4A (i) non-	4A (ii) non-	2A (i)	2A (ii)
arsenic	3.7	5.8	1.2	2.1
cadmium	0.1	0.1	0.0	0.0
chromium	220.0	260.0	76.0	89.0
copper	40.0	50.0	12.0	11.0
lead	0.0	0.4	0.0	0.0
nickel	11.0	16.0	1.9	4.5
zinc	5.3	5.5	5.0	5.9
Heavy metal	3B (i)	3B (ii)	4A (i)	4A (ii)
arsenic	2.6	2.5	1.1	2.1
cadmium	0.0	0.0	0.0	0.0
chromium	35.0	37.0	59.0	66.0
copper	7.4	10.0	10.0	9.2
lead	0.1	0.3	0.0	0.0
nickel	0.0	1.7	0.0	3.4
zinc	4.6	8.2	6.4	11.0

Appendix B: Burke St pavement construction proposal (Fulton Hogan)

Section: Burke Street
Client: City Care Christchurch
Researchers: NIWA – Dr. Craig Depree
Fulton Hogan – Thorsten Frobel
Date: February 2008



Report tracking:

		Rev. A	Rev. B	Rev. C
Prepared and designed:	Thorsten Fröbel Pavements Engineer Fulton Hogan	12 Nov '07	08 Feb '08	
Co-Researcher Review	Dr. Craig Depree Researcher NIWA			
Steering Group Review:	Dr. Bryan Pidwerbesky General Manager – Technical Fulton Hogan			

1 Introduction

This report covers the feasibility of using the foamed bitumen recycling technology as a structural requirement to carry the future traffic load on a 250m section of Burke Street located west of Antiqua Street. This is the first step that is required for the LTNZ-funded research project to investigate the encapsulation of PAH-contaminated pavement materials using foamed bitumen/cement in situ stabilisation.

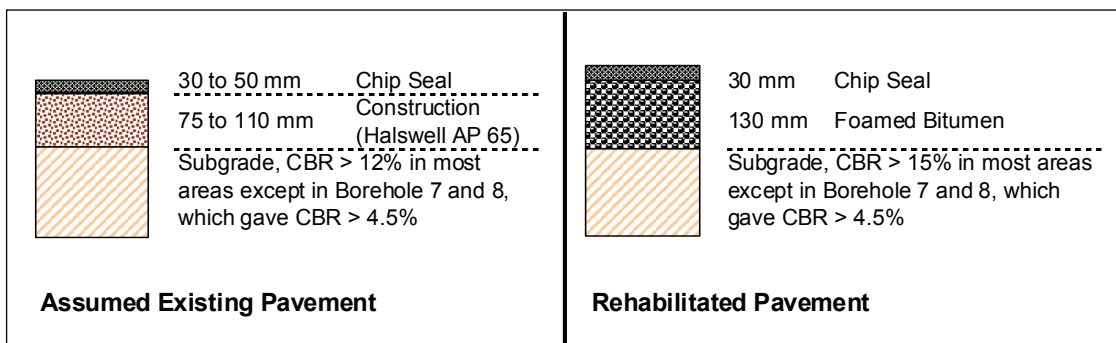
2 Available Information

Information supplied by Christchurch City Council (via City Care)

Christchurch City Council supplied a Borehole report for Burke Street, from Antiqua Street to House no.53.

From this initial pavement investigation the existing pavement has an average of 110mm of pavement construction on a sandy silt subgrade. The CBR along the majority of the site is > 15%, with only testpits 7 and 8, which are 190m and 213m from Antiqua Street, having a lower CBR of 4.5%. The AADT on this road is assumed/measured at 500vpd, which with a geometric growth of 0.5% and a 3% HCVs content gives a 25-year design standard axle repetition for the subgrade strain criteria of 90 thousand ESALs. From these parameters a rehabilitation option, using foamed bitumen is shown in figure B.1.

Figure B.1 Pavement structures before and after rehabilitation



From the pavement investigations, a total available recycling depth, which consists of the chip seal surface plus the construction of unknown quality, is 130mm. This is supported by a variable subgrade with a scalar determined CBR of between 4% and 15%.

3 Laboratory-based mix design testing

The seal and basecourse aggregates were sampled by City Care and freighted to Fulton Hogan's laboratory based in Hamilton, who has the foamed bitumen laboratory equipment. The results of the mix design are summarised on page 77.

The laboratory testing indicated very high dry and soaked indirect strengths at 1.0 and 2.0% foamed bitumen. In fact these high strengths should be avoided as the treated material will behave as a cemented layer, which attracts tensile fatigue, instead of a foamed bitumen modified layer that relies on the stress dependency of the granular material but immobilises the fines and thus avoids rutting by loss of fines, which is the primary purpose of the foamed bitumen.

The PI of the basecourse material was found to be between 10 and 15. Our first instinct, based on the ITS results, would be to lower the cement content to 1.0%. However, due to the moderate plasticity, the extra 0.5% cement will provide some security against shrinkage and swelling.

Therefore, we recommended a binder combination as 3% foamed bitumen + 1.5% cement.

4 Pavement design

The parameters of the pavement as detailed above have been analysed using the guidelines as set out in the *Austrroads pavement design guide* together with the *New Zealand supplement*.

Table B.1 Demonstrates the sensitivity of the varying subgrade, using the 800 MPa for the foamed bitumen treated layer, as recommended by the NZ supplement

Subgrade CBR [%]	FB thickness [mm]	Cumulative damage factor (CDF) [-]	Design traffic [thousand]	Service life [years]
15	100	0.264	>> 300	>> 25
10	100	2.0	45	10
10	130	2.0	45	10
4	130	72	< 1	< 1

As can be seen from the sensitivity study the subgrade CBR and existing pavement thickness has a large impact on achieving the required design life of 25 years or to carry the required design traffic of 90,000 ESAL.

The Benkelman beam deflections gave an average deflection of 3.5mm, which highlights the poor structural capacity of this pavement

5 Conclusion and recommendations

From the available information we feel that from chainage 115 to chainage 210 the subgrade support and available pavement poses a high risk to early failure and is therefore not an ideal area for foamed bitumen stabilisation. From Antiqua Street to chainage 115 the subgrade appears to have a reasonable CBR > 15% and the beam results are relatively lower than in the section discussed above.

We therefore recommend that, for the sake of researching the encapsulation of the PAH contaminated materials, only the first 115m is rehabilitated using the foamed bitumen technology.

Fulton Hogan has provided this pavement design report, based on the information given by City Care, in addition to following the guidelines used by Transit New Zealand and the industry. We therefore give no extended warranty for this work should it go ahead.

This report was compiled in good faith by:

Thorsten Fröbel CP Eng, Int PE MIPENZ
Pavements Engineer Fulton Hogan – Technical Services

Appendix

Pavement investigations

The eight test pits (TP) done by City Care reveal the following pavement structures:

Figure B.2 Schematic of the testpit investigations (by Citycare) of the existing pavement (red line at 130 mm depth indicating assumed pavement depth for design)

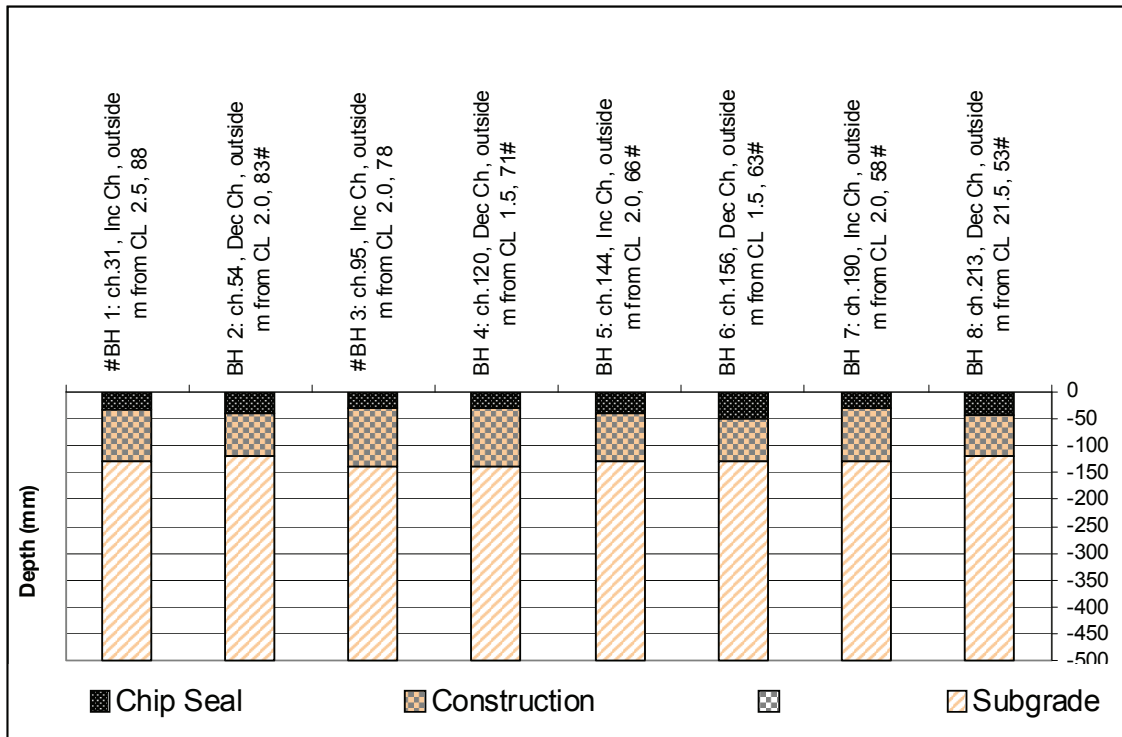
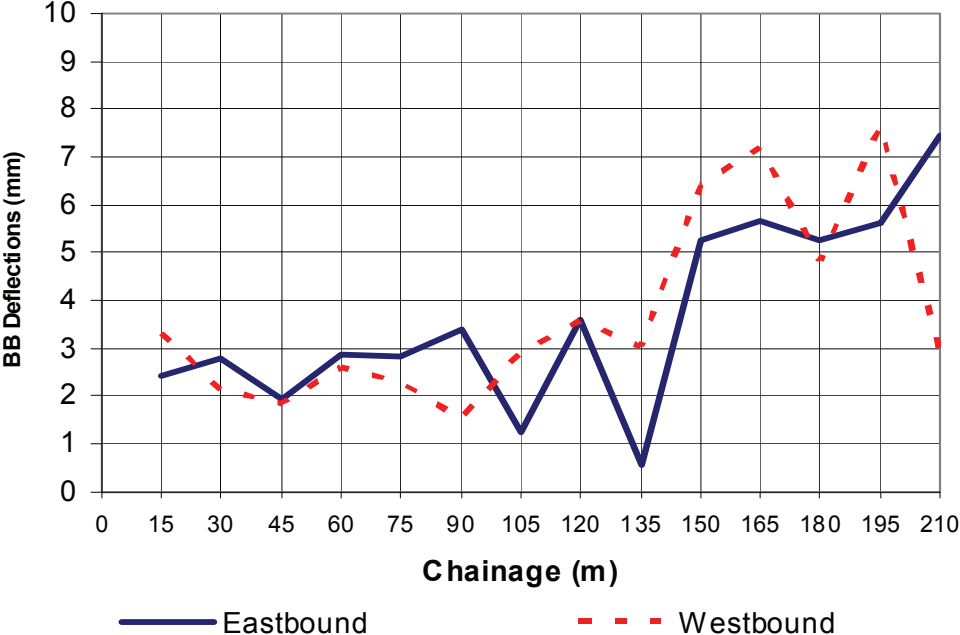


Table B.2 Pavement depths from the pavement investigations

BH 1: ch.31, Inc Ch, outside # 88, 2.5 m from CL	-35	-95
BH 2: ch.54, Dec Ch, outside # 83, 2.0 m from CL	-40	-80
BH 3: ch.95, Inc Ch, outside # 78, 2.0 m from CL	-30	-110
BH 4: ch.120, Dec Ch, outside # 71, 1.5 m from CL	-30	-110
BH 5: ch.144, Inc Ch, outside # 66, 2.0 m from CL	-40	-90
BH 6: ch.156, Dec Ch, outside # 63, 1.5 m from CL	-50	-80
BH 7: ch.190, Inc Ch, outside # 58, 2.0 m from CL	-30	-100
BH 8: ch.213, Dec Ch, outside # 53, 21.5 m from CL	-45	-75

Further Benkelman beam testing was carried out by City Care, with the following results:



25-year design load assumptions and calculations

Traffic Data

1st year:	2007
Current volume, AADT (vpd)	500 Annual Average Daily Traffic
Direction Factor, DF	0.5 Proportion of 2 way AADT travelling in design lane = 0.5 for a two-lane, two-way road with 50/50 directional split
%HV:	3.0% Heavy Vehicle content
N per HVAG	1.500 Average number of axle groups per heavy commercial vehicle. Presumptive value for NZ State Highways = 2.4
LDF	1 Lane Distribution factor - where there is unequal traffic distributed between the lanes of multilane roads
Design Life (years), y	25 Years
Traffic Growth Rate, i	0.5%

Damage Index Parameter

Damage Type	Damage Index Parameter = (SAR _m / ESA) x (ESA / HVAG)	SAR _m / ESA
Overall damage (ESA/HVAG)	0.6	
Asphalt fatigue	0.6	1
Subgrade rutting	0.8	1.2
Cemented layer fatigue	2.16	3.6

$$DSAR_m = [SAR_m / ESA \times ESA / HVAG] \times AADT \times DF \times \%HV \times N_{/HVAG} \times LDF \times CGF \times 365$$

Geometric Growth - Austroads Pavement Design Guide

CGF (Geometric) 26.559 Geometric Cumulative Growth Factor = $((1+i)^y - 1)/i$

	DIP	DSAR
Asphalt fatigue	0.6	6.544E+04
Subgrade rutting	0.8	8.725E+04
Cemented layer fatigue	2.16	2.356E+05

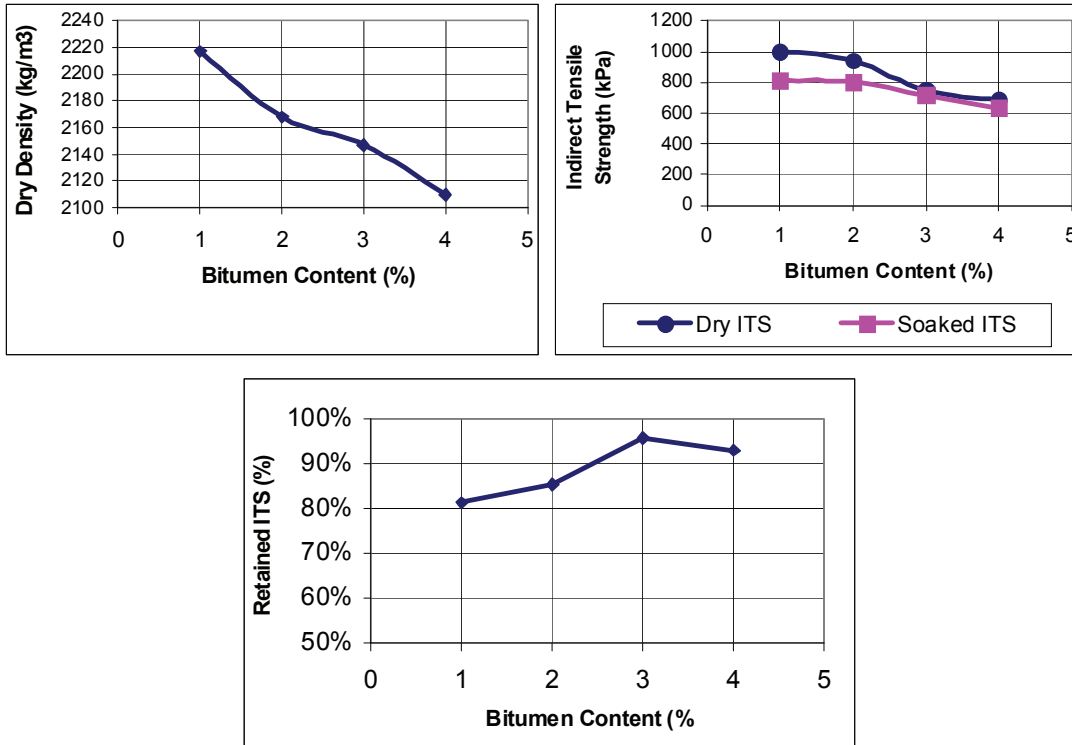
Arithmetic Growth - Transfund Project Evaluation Manual

CGF (Arithmetic) 26.5 Arithmetic Cumulative Growth Factor = $y + ((y/2) * (y-1) * i)$

	DIP	DSAR
Asphalt fatigue	0.6	6.529E+04
Subgrade rutting	0.8	8.705E+04
Cemented layer fatigue	2.16	2.350E+05

Laboratory results

The laboratory test results are as follows:



The laboratory testing indicated very high dry and soaked indirect strengths at 1% and 2% foamed bitumen. In fact these high strengths should be avoided as the treated material will actually behave as a cemented layer, which attracts tensile fatigue, instead of a foamed bitumen modified layer that relies on the stress dependency of the granular material but looks up the fines and thus avoids rutting by loss of fines, which is the primary purpose of the foamed bitumen.

The PI of the basecourse material was found to be between 10 and 15. Our first instinct, based on the ITS results, would be to lower the cement content to 1.0%. However, due to the moderate plasticity, the extra 0.5% cement will provide some security against shrinkage and swelling.

Therefore, we recommended a binder combination as **3% foamed bitumen + 1.5% cement**.

