Identifying Sensitive Receiving Environments at Risk from Road Runoff

Volume II - Appendices

Land Transport New Zealand
Research Report 315

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Land Transport New Zealand Research Report 315

*ISBN 0-478-28727-5
**ISSN 1177-0600

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Gardiner, L R & Armstrong, W. Identifying Sensitive Receiving Environments at Risk from Road Runoff, Volume II – Appendices, 2007. *Land Transport NZ Research Report No 315.* 108pp.

Keywords: contaminant, depositional environment, Geographic Information System, model, pathway, particulate, pollutant load, receptor, risk assessment, road runoff, screening tool, sensitive receiving environment, source, stormwater

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Acknowledgements

The authors wish to acknowledge Land Transport New Zealand for funding this research project. Kylie Jamieson and David Annan of MWH New Zealand Ltd prepared the GIS model and Karen O'Reilly developed the vehicle contaminant load model as part of this research contract. The Steering Group members are thanked for their contributions.

Abbreviations and Acronyms

AADT	Annual Average Daily Traffic volume
ADT	Average Daily Traffic
ANZECC	The Australian and New Zealand Environment and Conservation Council
Cu	Copper
CU	Central Urban
g	Gram
gis	Geographic Information System
HCV	Heavy Commercial Vehicle (>=3.5 tonnes)
km	Kilometre
Land Transport NZ	Land Transport New Zealand
LCV	Light Commercial Vehicle (<3.5 tonnes)
LoS	Level of Service
mg	Milligram [1/1000 g]
MO	Motorway
MoT	Ministry of Transport
ng	Nanogram [1/10^9 g]
PAH	Polycyclic aromatic hydrocarbons
Pf	Pathway factor
Ppm	Parts per million
RAMM	Road Assessment & Maintenance Management system
RE	Receiving environment
RH	Rural Highway
SRE	Sensitive receiving environment
SRf	Sensitivity rating factor
SS	Suspended Solids
SU	Suburban
TPH	Total Petroleum Hydrocarbons
Transit	Transit New Zealand
Transfund	Transfund New Zealand
µg	Microgram [1/10^6 g]
VCLM	Vehicle contaminant load model
VFEM	Vehicle Fleet Emissions Model
VKT	Vehicle Kilometres Travelled [AADT x Road Length]
VPD	Vehicles per Day
Zn	Zinc

Contents

Appendix A: Literature review of receiving environment sensitivity

A1. Introduction

A1.1 Purpose

Receiving environments include rivers, lakes, wetlands, estuaries, harbours and the open coastline that will exhibit varying degrees of sensitivity depending on their bio-physical characteristics, the uses that are made of them, and the values attached to them.

The primary aim of the literature review was to identify all of the factors relevant to assessment of receiving environment sensitivity and, in this respect, a logical starting point was a review of previous attempts to classify receiving environments according to their degree of sensitivity to receipt of contaminants.

The literature review commenced with an Internet search for relevant overseas publications[3]. However, this yielded little in the way of useful information and it appears that, in many respects, New Zealand researchers and resource managers are abreast of, if not leading, their overseas counterparts in this area. Consequently this paper focuses on New Zealand research and receiving environment classification systems.

A1.2 Content

This appendix contains seven sections as follows:

- Section 1: Introduction
- Section 2: Processes affecting contaminants in road runoff
- Section 3: The types of receiving environments and their physical characteristics
- Section 4: Methods for classifying receiving environments
- Section 5: Determinants of receiving environment sensitivity
- Section 6: Conclusions and recommendations
- Section 6: References

Recommendations from the literature review have been taken forward in the SRE rating framework and are taken forward in Section 3 of the main report.

A2. Processes affecting contaminants in road runoff

The nature and primary sources of vehicle-derived contaminants in road runoff are heavy metals (notably zinc from tyre wear and copper from brake pad abrasion), and hydrocarbons, notably Polycyclic Aromatic Hydrocarbons (PAH), from vehicle exhausts and lubricating oil.

For the present purposes, it is important to consider the way in which contaminants are transported in runoff and their fate when they reach freshwater or marine receiving environments.

Heavy metals and other road runoff contaminants can be added to the environment by being chemically or physically bound to sediment particles or as dissolved matter.

The conceptual picture emerging from studies undertaken in New Zealand and overseas is that ‘at source’ a high proportion of some contaminants (for instance zinc) may be present in the dissolved form. However, as it is carried through the drainage network, the dissolved fraction decreases as contaminants adsorb to particles (Timperley 2003). Furthermore, a large proportion of stormwater particulates are silt-sized or greater (Williamson, 1993; Leersnyder, 1993), such particles settling quickly in a suitable receiving environment.

The fate of particulate-associated contaminants is driven mainly by hydraulic processes in freshwater and by fresh-saline water interactions in an estuary. The latter includes physico-chemical processes at the fresh-saline water boundary, which may result in the coagulation of fine particulate material (Moncrieff and Kennedy 2004). In an estuary, most of the copper and zinc is attached to particulate matter and is incorporated into sand and mudflat sediments (ARC 2004).

The tendency of contaminants to adsorb to particulate material suggests that receiving environments which are depositional, that is where fine sediments settle (as evidenced by a soft silty or muddy substrate), will accumulate contaminants and will be most susceptible to adverse effects on benthic organisms.

Where stormwater is discharged into a dispersive receiving environment, such as a fast flowing stream or river, the bulk of the particulate material may be flushed downstream rather than accumulate on the stream bed. In such cases, consideration should be given to the downstream ‘ultimate receiving environment’ where the bulk of the particulate matter settles out (which may, for example, be a low gradient reach of river, lake or estuary).

Because contaminants accumulate along with sediments in depositional receiving environments, the effects of the discharge on sediment quality, and hence benthic organisms, are often the issues of primary concern[1].

The processes governing the fate and transport of contaminants in road runoff are explored more fully in Appendix B.

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[1] Food chain effects via bio-accumulation can also be important as benthic organisms are often at or near the base of the food chain.

A3. Types of receiving environments and their physical characteristics

A3.1 Introduction

Stormwater runoff from the New Zealand road network discharges directly to both freshwater and marine waters. Receiving environments can be divided into eight ecosystem types (Figure A3.1). The physical characteristics of each of these different types of receiving environment influence the way in which contaminant inputs are assimilated and therefore their sensitivity.

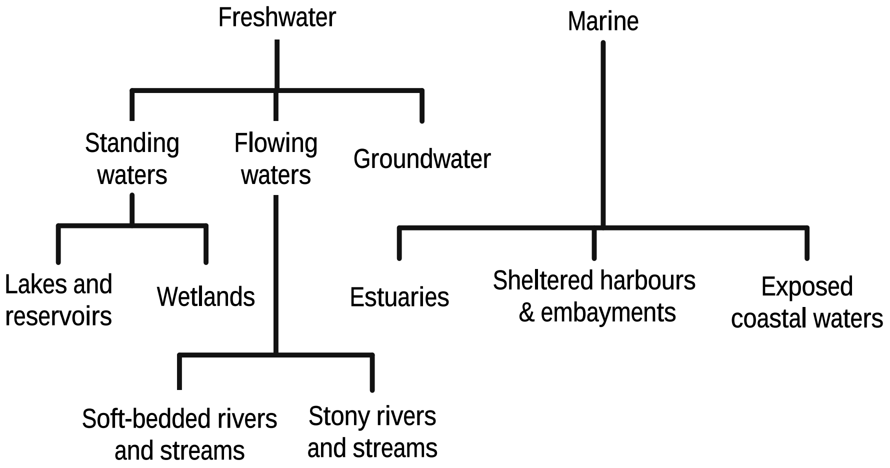

Figure A3.1 Receiving environment ecosystems.

It is important to consider the physical characteristics of each of these different types of receiving environment and the way in which these characteristics potentially affect their sensitivity or assimilative capacity.

A3.2 Physical characteristics of receiving environments

A3.2.1 Lakes and reservoirs

The two key processes that influence stormwater in the receiving water are dispersion/dilution of the discharge plume and *settlement* of particulate matter. Lakes and reservoirs typically provide a low energy depositional environment in which dispersion/dilution is limited by an absence of strong water currents, and large particles (sand and gravel) settle by gravity immediately adjacent to the discharge location. Finer particles may move away from the discharge location before settling, and may subsequently be re-suspended by wave action generated by strong winds.

The low energy depositional environment may result in a contaminant build-up in sediments at the point of discharge and, for small lakes with low natural sediment inputs, may result in a general increase in contaminant concentrations across the lake. This pattern of contamination has been observed in small lakes with a fully urbanised catchment, such as Lake Rotoroa in Hamilton (Williamson 2000).

To the extent that they form depositional environments, lakes and reservoirs are moderately to highly vulnerable to adverse effects derived from stormwater runoff.

A3.2.2 Wetlands

A definition of wetland for New Zealand purposes is provided in the Resource Management Act (1991):

"Wetland" includes permanently or intermittently wet areas, shallow water, and land water margins that support a natural ecosystem of plants and animals that are adapted to wet conditions.

The major difference between lakes and wetlands, from a scientific perspective, is depth. Wetlands are shallow water bodies, often with light penetration to the bed, while lakes are far deeper, resulting in the presence of both euphotic and profundal zones. These differences in depth and light penetration may result in the presence of aquatic macrophytes across the entire bed of a wetland while macrophyte communities in lakes are usually restricted to the shallow littoral regions.

Like lakes and reservoirs, wetlands provide a low energy depositional environment in which sediments and associated contaminants tend to accumulate. Indeed, the biofilms covering aquatic plants and other surfaces in wetlands can be particularly effective at trapping fine particulate matter from urban stormwater. This is one of the reasons that constructed wetlands or macrophyte ponds are commonly used to provide the final 'polish' in stormwater and wastewater treatment processes (Timperley et al. 2001).

Wetlands are dynamic systems with water levels that may change throughout the year from completely dry to flooded, and with plant and animal communities changing in response. This dynamic nature, and the wide variety of wetland types (bog, fen, marsh, saltmarsh, etc), makes it difficult to predict the way in which contaminant inputs will be assimilated and the degree of risk they will present.

In general, it appears that wetlands are moderately to highly vulnerable to adverse effects derived from road runoff.

A3.2.3 Soft-bedded rivers and streams

Soft-bedded rivers and streams in New Zealand are typically low relief watercourses with relatively low water velocities, which at most times generate insufficient energy to transport away all of the sediment inputs arriving from the catchment. This results in a gradual build-up of bed sediments that, over time, may be balanced by occasional flood flows which restore the base bed level by transporting accumulated sediments and contaminants downstream.

Depending on the ratio of stormwater sediment to uncontaminated sediment, elements such as zinc may accumulate in settlement zones and potentially exert an adverse effect on benthic organisms (e.g. Timperley 2000).

Many urban streams do not receive their normal supply of sediments and are dominated by contaminated sediment inputs derived from roadways and other urban sources. Furthermore, in many urban environments, stormwater flow could make up all of the flow in the water body resulting in minimal dilution and potentially high (but transient) concentrations of contaminants in the water column.

Timperley (2000) identified two pathways for adverse effects on aquatic life from chemical contamination. One is the direct interaction of free, dissolved contaminants, such as the zinc or copper ion, with animals' gills. This can occur in the water column and in sediment interstitial waters. The other pathway is through the incidental ingestion of contaminated particles, usually because these are mixed with normal food particles. A portion of the contaminants attached to particles is converted to dissolved forms in the animals' digestive system.

Soft-bedded streams are moderately to highly vulnerable to adverse effects derived from road runoff due to their tendency to trap and accumulate contaminants and, in the case of smaller urban streams, the limited capacity to dilute contaminated stormwater inflows.

A3.2.4 Stony rivers and streams

Stony-bedded rivers and streams are those higher gradient watercourses that generate sufficient energy to transport sediments downstream. The hydraulic processes are predominantly dispersive rather than depositional. Nevertheless, as described above for soft-bedded streams, adverse effects on aquatic life may occur either by direct interaction of dissolved contaminants with animals gills, or through the incidental ingestion of contaminated particles which may be trapped on biofilms, which are grazed by some macro-invertebrates.

In a study of urban streams in Auckland, Hamilton and Christchurch, Timperley (2000) found that zinc was the most significant dissolved metal, with chronic (base flow) levels possibly affecting up to 15% of aquatic organisms that could inhabit urban streams. This study indicated that transient peak concentrations, which occur in the early stage of a rainstorm, place additional stress on stream ecosystems but the magnitude of the effect was not quantified. Concentrations of copper, lead and zinc in fine suspended particulate matter were found to be very high at most sites, suggesting that ingestion of the particulate matter by grazing animals would greatly increase their dietary exposure to these metals.

The vulnerability of stony streams to adverse effects derived from road runoff would be in the low to moderate range, depending on the size of the stream and its capacity to dilute urban stormwater inflows.

A3.2.5 Groundwater

Stormwater discharging into a roadside drain, grass swale, rain garden or other form of ground soakage may seep directly into the shallow groundwater. Its passage through surface soils tends to filter out particulate material, which is retained near the surface of the soil profile. Retention of dissolved constituents would depend on the soil matrix.

Stormwater entering the groundwater can therefore be expected to be substantially free of particulate material but may not necessarily have reduced concentrations of dissolved contaminants.

Groundwater would not normally be considered to be vulnerable to adverse effects derived from stormwater discharges.

A3.2.6 Estuaries

Contaminants primarily adsorb to fine particulate matter in stormwater discharges (Williamson, 1993; Leersnyder, 1993). Upon arrival at the estuary, the coarser particles settle by gravity because of the large drop in water velocity. Finer particles are flocculated and the resultant larger particles settle to the bed. Therefore the immediate fate of a large proportion of the contaminants, after entering the estuary, is deposition by settling in the upper reaches of the estuary.

Some particle and dissolved contaminants will be carried through the estuary, particularly during large storms, and especially during low tide when storm flows are carried right down the estuary in central incised channels. Dissolved contaminants in stormwater tend to adsorb to particles in the estuary (ARC 2004).

Recent monitoring of sediment quality in Auckland’s urbanised estuaries by the Auckland Regional Council shows that at some 29% of estuarine sampling sites, zinc exceeds the initial threshold of Environmental Response Criteria (ERC) set in the Proposed Regional Plan. Levels of copper, lead and PAH also exceed the initial threshold level at a number of locations. The highest contaminant levels were found in settling zones of catchments with the longest history of urbanisation.

Not surprisingly, most settling zones and outer zones away from the main urban areas, that have catchments predominantly in rural land use, have low concentrations of these contaminants (ARC 2004, Diffuse Sources 2004). Trend analysis of the ARC Long Term Baseline monitoring programme demonstrates that zinc and copper concentrations are clearly increasing at many sites, while lead concentrations are decreasing[2]. Stratigraphic information from cores taken in urban estuaries confirms the increase in copper, lead and zinc with change in land use from rural to urban. It also confirms the more recent decrease in lead (ARC 1994, Swales et al 2002, Williamson et al 2003).

As is the case for freshwater systems, aquatic life in estuaries can be affected by stormwater contaminants via two potential pathways. One is the direct interaction of dissolved contaminants with animals’ gills, either in the water column or in sediment

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[2] Decreasing concentrations of lead are thought to reflect the removal of lead from petrol in the early 1990s.

interstitial waters. The other pathway is through the incidental ingestion of contaminated particles mixed with normal food items.

Estuaries, particularly those within sheltered harbours, are highly vulnerable to adverse effects derived from stormwater discharges.

A3.2.7 Sheltered harbours and embayments

Stormwater discharging to a sheltered harbour or embayment is subject to many of the processes described above for estuaries. Several studies have recorded the build-up of contaminants in sheltered inner harbours and some have identified adverse effects on benthic ecology.

For example MWH (2003) records a build-up of contaminants (mainly zinc and copper) and decreased species diversity and dominance of optimistic species in the vicinity of stormwater outfalls in inner Wellington Harbour.

Sheltered harbours and embayments, being deposition environments, are highly vulnerable to adverse effects derived from stormwater discharges.

A3.2.8 Exposed coastal waters

Exposed coastal waters are high-energy ecosystems in which the discharge plume and associated particulate material is rapidly dispersed by turbulence and currents driven by wind, wave, and tidal action as well as larger-scale influences such as ocean currents.

Contaminants derived from road runoff therefore tend to be rapidly diluted and dispersed through coastal waters rather than accumulating around the outfall. Exposed coastal waters have a relatively low vulnerability to adverse effects from stormwater discharges.

A4 Methods for classifying receiving environments

A4.1 Introduction

Methods for classifying receiving environments in New Zealand can be divided into two categories:

i) Those aimed at broad classification of ecosystems, or parts of ecosystems, on the basis of their physical or biological characteristics ('eco-types').

ii) Those involving attempts to classifying receiving environments according to the characteristics considered to relate more directly to their 'sensitivity', and hence their ability to assimilate contaminants without showing significant adverse effects.

A4.2 Eco-typing approaches to classification

Eco-typing involves the grouping of ecosystems or parts of them with similar physical and biological characteristics, the presumption being that water bodies with similar characteristics lend themselves to similar management treatments. This being the case, it is implicit that water bodies grouped into a specific "eco-type" have a similar degree of sensitivity to pollution.

A4.2.1 ANZECC (2000)

A review of ecosystem classification schemes in the ANZECC water quality guidelines (ANZECC 2000) found that three broad categories have emerged:

- Those based entirely on geography (e.g. inland, estuarine, coastal/marine).
- Those based on climate (e.g. tropical, temperate, arid).
- Those based on geography and/or climate coupled with a consideration of key physical and biological factors.

The majority of the older classification systems (e.g., Hughes & Larsen 1988; Biggs et al. 1990) are based on physical geography and use manual classification techniques to draw visible ecological boundaries onto maps. More recent classifications are numerically-based using computer programs to sort climate, landform and soils data to group areas containing ecosystems of similar type.

The ANZECC (2000) water quality guidelines classify ecosystems into six broad groups (estuarine, coastal marine, lakes and reservoirs, wetlands, upland rivers & stream and lowland rivers & streams), and recognises three 'ecosystem conditions' based on their degree of modification. The three ecosystem conditions are:

i) *High conservation/ecological value systems:* Effectively unmodified or other highly valued ecosystems, typically (but not always) occurring in national parks, conservation reserves or in remote and inaccessible locations. The ecological integrity is regarded as intact.

ii) *Slightly-to-moderately disturbed systems:* Ecosystems in which aquatic biological diversity may have been adversely affected to a relatively small but measurable degree by human activity. The biological communities remain in a healthy condition and ecological integrity is largely retained. Slightly-to-moderately disturbed systems could include rural streams receiving runoff from farmland, or marine ecosystems lying immediately adjacent to metropolitan areas.

iii) *Highly disturbed systems:* These are measurably degraded ecosystems of lower ecological value. Examples of highly disturbed systems would be some shipping ports and sections of harbours serving coastal cities, urban streams receiving road and stormwater runoff, or rural streams receiving runoff from intensive horticulture.

The third ecosystem condition recognises that degraded aquatic ecosystems still retain, or after rehabilitation may have, ecological conservation values, but for practical reasons it may not be feasible to return them to a slightly-to-moderately disturbed condition.

ANZECC (2000) recommends that these levels of ecosystem condition be used as a framework to decide upon an appropriate level of protection. Key stakeholders in a region would normally be expected to decide upon an appropriate level of protection through determination of management goals and based on the community's long term desires for the ecosystem.

The philosophy behind selecting a level of protection, inherent in the water quality guidelines, is either (i) maintain the existing ecosystem condition, or (ii) enhance a modified ecosystem by targeting the most appropriate condition level.

A4.2.2 NZWERF (2002)

The New Zealand Water and Environment Research Federation (NZWERF), in The New Zealand Municipal Wastewater Monitoring Guidelines 2002, adopts a risk-based approach as a basis for developing receiving environment monitoring programmes. The approach, which is said to build on the ANZECC ecosystem classification (above), aims to ensure that a monitoring programme devised for any given situation, amongst other things, reflects the true risks faced by the receiving environment.

The process used in the Guidelines for the risk-based development of a monitoring programme is termed the HIAMP process (Hazard Identification, Analysis and Monitoring Plan). The first step of the HIAMP process involves characterisation of the discharge, the receiving environment and community values. In Step 2 this information is fed into the Risk Analysis (identification of 'hazards' associated with the discharge and/or the receiving environment, and assessment of the potential level of impact associated with each hazard) and hence the design of an appropriate monitoring plan in Step 3.

Classification of the receiving environment is thus a fundamental step in the HIAMP process. As NZWERF (2002) notes, at p49:

> *Receiving environment classification can take many forms and can be very complex or very general, depending on the desired outcome. For example, physical factors such as climate, geography and biology are often used to 'ecotype' on environment, but social and cultural factors are also important and might be incorporated under certain circumstances. In general, characterisation of the receiving environment allows for the creation of groups or types of environments, which will react in a different fashion when exposed to a wastewater discharge.*

Under the NZWERF approach, receiving environment characteristics are divided into two primary categories: (i) type of environment (e.g. lake or estuary) and (ii) characteristics within each environment that affect the extent to which wastewater components will be assimilated (called ‘assimilative capacity’).

The types of receiving environment recognised by NZWERF are:

i. Lake/Reservoir
ii. River/Stream (where wastewater input is <50% base flow)
iii. River/Stream (where wastewater input is >50% base flow)
iv. Estuary
v. Harbours and sheltered embayments
vi. Nearshore marine (shoreline)
vii. Offshore marine
viii. Groundwater

The assimilative capacity characteristics of the receiving environment recognised by NZWERF are:

i. Dilution
ii. Substrate
iii. Enrichment status
iv. Sensitivity of ecological values
v. Significant other inputs to the environment
vi. Aesthetics
vii. Human health and safety (via contact recreation)
viii. Water supply (whether or not)
ix. Food gathering
x. Cultural or spiritual value

NZWERF then applies a rather complex matrix approach to assessing, for a given receiving environment, the hazards represented by the characteristics of the effluent and the receiving environment characteristics, and this information is then fed into the HIAMP model.

A4.2.3 Ministry for the Environment research

Over the last five years, New Zealand’s Ministry for the Environment has led a series of major research projects aimed at developing a numerically-based approach to environmental classification.

The outcome of this work includes the formulation of the ‘New Zealand River Environment Classification’ as described by Snelder et al. (2004) and the ‘New Zealand Marine Environment Classification’ as described by Snelder et al. (2005).

The New Zealand River Environment Classification (REC) is a spatial framework for regional scale environmental monitoring and reporting, environmental assessment and management. It is intended to assist with:

- organising empirical data,
- extrapolating data and information to locations with no data,
- stratifying variation in rivers so that monitoring sites can be selected and management activities can be prioritised, and
- summarising the characteristics of types of rivers so that management expectations and controls can be set that are justifiable and achievable.

The REC groups and classifies rivers (or parts of rivers) at six hierarchical levels, each corresponding to a controlling environmental factor. The factors, in order from largest spatial scale to smallest, are:

- climate,
- source-of-flow,
- geology,
- land cover,
- network position,
- valley landform.

The REC is provided as a GIS layer that can be displayed as a series of maps showing classes at each level of the REC hierarchy.

The 'New Zealand Marine Environment Classification' (MEC) covers both New Zealand's Exclusive Economic Zone and, at a higher level of resolution, the Hauraki Gulf region. The purpose of the classification is to provide spatial frameworks for structured and systematic management by subdividing the geographic domain into units having similar environmental and biological character.

The MEC may be utilised in a variety of applications including:

- Defining management units that will be subject to similar objectives, policies and methods,
- Transferring knowledge of processes and values to other areas on the basis of similarity,
- Predicting the potential impacts of events and resource uses based on ecosystem susceptibility,
- Identifying priorities for protection (e.g. which parts of the environment should be included in marine protected areas), and
- Structuring monitoring programmes to ensure they represent all environmental types, and providing a context for reporting state of the environment information.

Both the REC and MEC classification systems provide managers with a useful framework for broad scale environmental and conservation management. However the full utility, and indeed limitations of the classifications, will only become clear as the classifications are applied to management issues.

An obvious limitation in respect of road runoff risk assessment is that neither the REC nor the MEC classification systems address estuaries, which are known to be particularly vulnerable to stormwater contamination.

A4.3 Examples in New Zealand of ranking the sensitivity of receiving environments for management purposes

Methodologies for identifying and ranking sensitive receiving environments have been developed in New Zealand by local and regional authorities for a variety of purposes. Examples (discussed below) relating to the discharge of stormwater include:

- Studies commissioned by the Waitakere City Council during the preparation of their '*Comprehensive Urban Stormwater Management Strategy and Action Plan*' (September 2000).

- The Hawke's Bay Regional Council's Stormwater Contaminant Planning Maps (GHD July 2005, in draft).

A4.3.1 Waitakere City Council

The Waitakere City Council Urban Stormwater Management Strategy identified 33 stormwater management units. They form the broad geographic basis on which stormwater is managed in Waitakere City and are mainly defined on the basis of land use and catchment boundaries. The stormwater management units were grouped together on the basis of:

- a sensitivity ranking of their coastal receiving environments,
- their ecological values,
- their community use.

The marine and estuarine receiving environments of the stormwater management units are distinguished on the basis of their geographical location and tidal flushing characteristics. The sensitivity ranking of these receiving environments were derived in consultation with the Auckland Regional Council by adding together factors representing an ecological value of the water body and its vulnerability to degradation.

For example, the Upper Waitemata Harbour and enclosed Whau River mouths are depositional, low energy, environments where fine sediment settles. Because they have slower flushing rates than the middle Waitemata Harbour, they are more vulnerable to water and sediment quality degradation, and so receive the highest vulnerability rating of 3.

Table A4.1 shows the sensitivity ranking for Upper Waitemata Harbour is 4 + 3 = 7, and for Whau Estuary is 2 + 3 = 5.

Table A4.1 Sensitivity ranking of receiving environments (Waitakere City Council).

Estuarine/ marine receiving environment	Stormwater management units	Ecological value of receiving environment [(a)]	Vulnerability of receiving environment to degradation [(b)]	Sensitivity ranking of receiving environment [(c)]
Upper Waitamata Harbour	(27) Whenuapai (25) Herald Island (26) Redhills	4	3	7
Whau Estuary	(5) Wairau Creek (4) New Lynn East (3) Rewarewa etc.	2	3	5

Notes: (a) rating from 1- 5, with 5 being the highest value; (b) rating from 1-3, with 3 being the most vulnerable; (c) ranking from 2 to 8, with 8 being the most sensitive

The receiving environment sensitivity rankings, together with a wide range of other issues (such as community use, flooding, erosion, land development potential), were weighted and combined into an overall prioritisation for the purpose of programming management plans and funding of capital works. In practice, the weighting given to receiving environment sensitivity meant that it had very little influence on the overall prioritising of capital works.

A4.3.2 Hawke's Bay Regional Council

The Hawke's Bay Regional Council (HBRC) has developed a methodology for the development of GIS information maps that classifies sensitive receiving environments and industry types to enable more effective management of industrial stormwater discharges, and to minimise effects on the environment (GHD 2005).

The primary classification is based on environment types, as shown in Table A4.2.

Table A4.2 Primary classification of environment type adopted by HBRC.

Category	Description	Environment Types
A	Sink environment, settling & accumulation area, low disturbance and redistribution levels – contaminants unlikely to be dispersed to other areas.	Estuary, lake or pond, wetland
B	Dynamic environment – moderate levels of disturbance and redistribution, some settlement areas and low-medium baseflow volume with less dilution potential.	Stream
C	Dynamic environment – moderate to high levels of disturbance and redistribution, limited settlement areas, high base flow volume with good dilution potential.	River
D	Dynamic environment – high levels of disturbance – contaminants likely to be dispersed within 24 hours.	Coast

The secondary classification (Table A4.3) takes into account the ecological value of the environment. The tertiary classification (Table A4.4) takes account of the ability of the environment to assimilate contaminants without major degradation.

Each area is to be classified according to the primary and secondary categories, and possibly the tertiary classification. The primary classification is presented as a coloured polygon on a GIS map, with the secondary and tertiary classification represented as a number allocated to the coloured shape.

Table A4.3 Secondary classification of environment type adopted by HBRC.

Category	Description
1	Classified by HBRC or DOC as important ecological areas (e.g. protected natural area, wildlife refuge, reserve, restoration site, etc).
2	Rare or keystone species or rare habitat thought to be present (not necessarily formally classified) or is a good representative example for the region.
3	Highly productive habitat, supports high biodiversity, acts as nursery habitat, or provides connection between important areas.
4	Site adds to the general regional ecology.
5	Site would benefit greatly from minor-moderate restoration works.

Table A4.4 Tertiary classification of environment type adopted by HBRC.

Category	Description
i	Considered already highly degraded and has minimal remaining assimilation capacity – significant cumulative affects – contaminants may begin 'overflowing' to environments.
ii	Considered already moderately degraded and has a limited assimilation capacity – cumulative effects occurring and expected to worsen.
iii	Considered to be degraded and has a diminished assimilation capacity, some cumulative effects occurring.
iv	Considered to have minimal degradation, has some assimilation capacity, no obvious cumulative effects occurring.

The industries within the region are classified on the basis of their use or production of hazardous substances, with the secondary classification based on proximity to sensitive receiving environments. The alpha-numeric classification system is used to create databases of geographical areas and the resultant planning maps are expected to be similar to land use planning maps in a district plan.

The objectives of the Hawke's Bay Regional Council initiative have a number of similarities with objectives of this project. The methodology for ranking sensitive receiving environments is particularly relevant and was taken into account in developing the ranking system in this project.

A5 Determinants of receiving environment sensitivity

A5.1 Introduction

From the literature, it is apparent that receiving environment sensitivity or vulnerability to adverse effects can be by determined by one or a combination of several factors including physical characteristics, ecological values, and the specific human uses or values associated with the water body in question.

This section summarises and identifies the key factors and briefly discusses how they influence sensitivity, drawing where appropriate from the material referred to above.

A5.2 Waterbody characteristics

The important physical characteristics of a receiving environment that influence its vulnerability to stormwater contaminants include:

- The dilution available and the rate at which mixing and dispersion occurs (governed by the size of the receiving water body relative to stormwater inflows, and receiving water velocity).
- The rate of sediment deposition (governed largely by the extent of water movement, including wave action and tidal/ocean currents).

These characteristics will influence the concentration of contaminants in the water column near the stormwater outfall and the extent to which stormwater particulates are transported away from the discharge point.

Low energy ‘depositional’ or ‘sink’ environments, with little water movement, are at greatest risk of build-up of contaminants in fine sediments, to levels representing a threat to benthic organisms. The levels of dissolved contaminants in the water column are generally low in depositional environments (where they rapidly adsorb to fine particulate matter). However, in some small urban streams, dissolved contaminants may represent a threat to aquatic organisms.

Generally speaking, high energy ‘dispersive’ environments with significant water and sediment movement (e.g. rapidly flowing rivers or open coastlines) are at low risk of adverse effects as contaminants are rapidly mixed[3], diluted and dispersed.

As a guide, receiving environments can therefore be grouped according to their risk from runoff as shown in Table A5.1.

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[3] The rate of which mixing occurs is governed largely by the velocity of the receiving water but the degree of turbulence can also be a significant factor.

Table A5.1 Risk-based grouping of receiving environments to runoff.

Depositional environments (high risk)	↔ **(moderate risk)**	**Dispersive environments (low risk)**
• Enclosed harbours, ports • Estuaries • Low gradient, slow-flowing streams with small base flow • Small lakes, reservoirs • Wetlands	• Semi-enclosed harbours and embayments • Large lakes • Moderate velocity rivers with medium base flow	• Open coastline • High gradient, fast-flowing rivers with large base flow

A5.3 Natural values present

A receiving water can be sensitive by virtue of the high natural or ecological values present. For example, a 'high' degree of sensitivity might be ascribed to receiving waters with:

- Rare, threatened or endangered species present,
- Communities with high species diversity,
- Presence of habitats or communities that are particularly sensitive to stormwater-related effects, or
- High conservation status (e.g. a water body identified as being of national or regional significance, or one which is within a reserve area).

A lower degree of sensitivity is indicated by receiving waters that are characterised by:

- Relative lack of biota, or
- An environment that is impoverished, homogenous or ubiquitous.

The assignment of a high priority to protecting significant ecological values is consistent with the RMA's emphasis on i) avoiding adverse effects on ecosystems, ii) safe-guarding life-support systems, iii) protecting significant indigenous vegetation and habitats of indigenous fauna, and iv) sustaining the potential of natural resources to meet the reasonably foreseeable needs of future generations.

At face value, it would see reasonable to afford a higher status to the ecological values rather than the human uses values associated with a given waterbody in any system for classifying the sensitivity of receiving environments. The reason for this is that humans are fundamentally dependant upon the `health' of the biosphere and ecological processes for their long term social and economic wellbeing. This aspect is given further consideration in Appendix C.

A5.4 Human uses and values

A receiving water may be sensitive by virtue of the human uses or values associated with it. For example, a 'high' degree of sensitivity might be ascribed to receiving waters:

- Used extensively for contact recreation,
- Of high aesthetic/recreational/tourism value,
- Of cultural value e.g. valued by Maori as a customary source of food, or for spiritual reasons, or
- Used as a drinking water source.

The objective of protecting significant human uses and values associated with water bodies is consistent with RMA requirements to i) avoid, remedy or mitigate adverse effects on the environment, ii) to recognise and provide for the relationship of Maori and their culture and traditions with their ancestral lands, water, waahi tapu and other taonga, and iii) to have particular regard to the maintenance and enhancement of amenity values and the quality of the environment.

A5.5 Existing degree of contamination or disturbance

The extent of contamination or disturbance already present in a receiving environment can be viewed in two quite different ways:

i. The environment is already degraded and therefore not so sensitive.

ii. It is more sensitive because maintaining or increasing the contaminant load could increase the stress on the ecosystem, increase the size of the impacted zone, and increase the potential for adverse effects.

The authors tentatively favour the latter view on the basis that it is consistent with the risk-based approach to assessing sensitivity and avoidance/remediation priorities, and because it is consistent with the requirement to take into account ‘cumulative’ effects under the RMA.

This aspect is further discussed in Section C6 of Appendix C.

A6 Conclusions and recommendations

A6.1 Conclusions

It is apparent from the literature review that receiving environment sensitivity or vulnerability to adverse effects can be determined by one or a combination of several factors. These include:

- The physical characteristics of the waterbody
  - The size of the water body (dispersal and dilution characteristics), or
  - Water movement (determines rates of mixing, dispersion rates and sediment deposition).

- The natural/ecological values associated with the waterbody
  - Rare/endangered species,

  - Rare/scientifically significant communities,
  - Communities with high species diversity,
  - Habitats/communities particularly sensitive to stormwater-related effects, or
  - High conservation status (e.g. a waterbody of national or regional significance, or one within a reserve area).

- Human uses and values associated with the waterbody
  - Contact recreation,
  - Aesthetic/recreation/tourism values,
  - Cultural values, or
  - Drinking water source.

- The existing degree of contamination or disturbance.

The available information indicates that water movement is a key, if not the primary factor, in determining the sensitivity of a receiving environment to stormwater inputs. This is because it determines whether or not sediments are deposited (rather than dispersed) and hence whether or not the concentrations of sediment-attached contaminants are able to accumulate to potentially harmful levels.

Assignment of a high priority to ecological criteria in assessing receiving environment sensitivity is consistent with the requirements of the RMA (see Section A5.3). The RMA also requires consideration to be given to human uses and values (Section A5.4).

The international literature search to date has yielded little in the way of useful intention pertaining to the ranking or classification of receiving environment sensitivity.

The approach taken by NZWERF (Section A4.2.2) has regard to a wide range of receiving environment ‘types’ and assimilative capacity characteristics. However the method is developed primarily for wastewater effluents[4] and it adopts a rather complex matrix approach to assessing, for a given receiving environment, the hazards represented by the characteristics of the effluent and receiving environment. This information is then fed into a model. It is understood that there has been limited ‘uptake’ of the NZWERF model by resource management practitioners, and it seems likely that this could reflect the complexity of the approach.

Both the Waitakere City Council and Hawke’s Bay Regional Council approaches have merit (see Section A4.4) and have been taken into account in developing the proposed receiving environment sensitivity rating system, as described in Section C3.5 and Appendix C.

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[4] Stormwater differs significantly from wastewater in that it generally has a lesser range of contaminants of potential concern, and a high proportion of the contaminants are associated with sediment particles, as outlined in section A2.

A6.2 Recommendations

On the basis of this literature review, the following considerations have been taken into account in developing the SRE screening methodology during Stage 2 of this Project:

- The identified 'sensitivity factors' including the physical characteristics, natural/ecological values and human uses and values of the waterbody.
- The possible merits of a simple visual approach to the classification and ranking of receiving environment sensitivity – as opposed to a complex, computer based approach.
- The primary role that water movement plays in determining whether or not a receiving environment is 'depositional' or 'dispersive' and hence its assimilative capacity/susceptibility to contaminant build-up in sediments.
- The important, but secondary, role that natural/ecological values play in determining the sensitivity of receiving environments.
- The important, but tertiary, role that human uses or values play in determining the sensitivity of a receiving environment.
- The desirability of factoring in the existing degree of contamination or disturbance, and other contaminant sources, when assessing the sensitivity of a receiving environment.

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Appendix B: Development of an SRE sensitivity rating system

B1. Introduction

B1.1 Purpose

Stage 2 involved developing a screening methodology for evaluating the sensitivity of different types of receiving environments. This included a `sensitivity rating system' based on key attributes i.e. receiving environment type, ecological value, and human use (including cultural) value, identified under Stage 1.

An important consideration in developing the methodology is to identify the contaminants of primary concern and to consider their chemical or physical state in the runoff (i.e. during transport from the road surface to receiving environments). This has a bearing on the fate of contaminants and hence the types of receiving environments most likely to be affected by road runoff.

This report draws on the findings of both the Stage 1 report and additional literature review in addressing these risk factors.

The proposed methodology for determining the sensitivity rating of receiving environments to road runoff is summarised in Section 3.5 and described more fully in Appendix C.

B1.2 CONTENT

Appendix B covers three topics:

- Nature, transport and fate of contaminants in road runoff,
- Effects of road runoff on receiving environments, and
- Conclusions on key attributes for the sensitivity rating system.

B2. The nature, transport and fate of contaminants in road runoff

B2.1 Contaminants of concern

Road runoff contains a potentially wide range of contaminants including heavy metals, organic compounds and sediments. Readers are referred to the Ministry of Transport research report (Kennedy 2003) for detailed information relating to the contaminants potentially present in road runoff and for a summary of existing knowledge relating to the concentration of contaminants in both urban stormwater and in runoff from roads and highways.

From the point of view of potential for adverse effects on the receiving environment, the key contaminants in road runoff, and their primary sources, are listed in Table B2.1.

Table B2.1 Key contaminants of concern in road runoff.

Contaminant	Sources
Zinc (Zn)	Tyres, brake pads, bitumen, roofs of buildings
Copper (Cu)	Brake pads, bitumen
Lead (Pb)	Brake pads, tyres
Hydrocarbons (including polycyclic aromatic hydrocarbons, PAHs)	Vehicle lubricant oil, exhausts
Particulates / sediments	Road surface, bitumen, tyres, road verges

Two points should be noted concerning the above table. Firstly, the compositions of contaminants in bitumen, tyres and brake pads (detailed in Kennedy et al. 2002, Kennedy 2003) are highly variable and can have a significant impact on the nature and variability of the contaminant load in road runoff. Second, although lead is still present in road runoff, its significance as a pollutant in road runoff has decreased markedly since its removal from petrol in New Zealand in 1996.

B2.2 Dissolved versus particulate contaminants

As discussed in Appendix A, heavy metals and other contaminants in road runoff can be transported as either dissolved species or by being chemically or physically bound to sediment particles (particulate contaminants).

Contaminants in the dissolved phase are biologically available (uptake by gills) and hence potentially toxic to aquatic life. Contaminants associated with particulates are insoluble and therefore not directly available to aquatic life via gill uptake. However they can exert toxic effects on macroinvertebrates by way of grazing on sediment particles trapped on ‘biofilms’ or through ingestion of contaminated sediment after it has been deposited. Sediment-associated contaminants can also revert to the dissolved phase (depending on sediment characteristics, e.g. pH, eH) and become biologically available following release to interstitial or overlying waters.

The proportion of an element or compound that is present in road runoff or in receiving water at a given location depends on the properties of the element or compound in question (e.g. its solubility) and the physico-chemical characteristics of the water, e.g. pH, organic matter or sediment content.

Studies undertaken overseas and in New Zealand indicate that ‘at source’ a high proportion of some contaminants (e.g. zinc) may be present in the dissolved form but, during passage through the drainage network, the dissolved fraction decreases as contaminants absorb to particles (Timperley 2003).

The New Zealand data (e.g. Timperley 2003) indicate that zinc is the most soluble (up to 40% of total metal in the soluble phase) of the key elements in stormwater. Copper appears to be moderately soluble (about 30%) and lead is the least soluble (<10%). A high proportion of the total concentration of low molecular weight PAH is present in the dissolved phase (Kennedy 2003).

The literature indicates that a high proportion of the total contaminant load arising from a road is typically associated with the solid or particulate fraction of a discharge. Particles are transported in suspension or in 'bed load' along the bottom of the stormwater pipe or stream channel.

In the UK, (Highways Agency 2000), estimates indicate that some 60-90% of the total copper is likely to be bound to the sediment fraction[5]. The solid fraction also contains over 90% of the inorganic lead and 56% of the cadmium (Kennedy 2003).

Timperley (2001) examined the suspended solids present in urban stormwater discharges from a variety of sources in Auckland, Hamilton and Christchurch. The results showed that suspended material can contain high concentrations of copper (e.g. median values of the order of 50-250 mg/kg), lead (50-400 mg/kg) and zinc (500-2500 mg/kg) with the concentration typically increasing from residential to commercial to industrial land uses.

In estuaries it is known that most of the total copper and zinc is attached to the particulate matter and it is thought that coagulation processes facilitate the incorporation of these particles into sand and mudflat sediments.

PAHs, which are derived from unburnt fuel and are potentially toxic, have a higher affinity for the sediment fraction than most other hydrocarbons (Ellis and Revitt 1991). High molecular weight PAHs are almost entirely present in the particulate phase (Kennedy 2003).

In the UK, up to 70% of the oil deposits deposited onto a road by moving vehicles becomes associated with the sediment fraction and may ultimately settle out on the bed of the receiving water (Highways Agency 2000).

B2.3 The fate of contaminants in road runoff

The fate of road transport derived contaminants is primarily dependent upon the chemical state of the contaminant (dissolved versus particulate, above) and the hydro-dynamics of the receiving environment at the point of discharge.

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[5] The proportion of dissolved vs. insoluble (bound) copper in stormwater runoff and/or in a receiving environment can be influenced by the amount of dissolved organic matter (DOM) present, DOM promoting complexing of copper.

B2.3.1 Fate of dissolved contaminants

A portion of the dissolved metals and organic compounds in the runoff will be subject to biological uptake, via animal gills, in the immediate receiving environment or in downstream receiving environments.

However, there will be a general tendency for dissolved contaminants to be diluted and dispersed in the receiving environment. The rate at which this occurs will vary from situation to situation depending on the volume of the discharge and the size and mixing characteristics of the receiving water.

Where the energy of the receiving environment is high and large volumes of water are available for dilution (e.g. high velocity turbulent stream, or the open coast) contaminants will disperse rapidly. There is, however, some potential for the build-up of dissolved contaminants in the water column in low energy environments, particularly above contaminated sediments.

B2.3.2 Fate of contaminants associated with particulate fraction

In contrast to dissolved contaminants, particulate-associated contaminants will tend to settle and accumulate in the nearest (downstream or downcurrent) depositional receiving environment. Solids in road runoff generally have a low particle size (predominantly <100μ diameter) and relatively high density.

If the stormwater discharge is to a low energy section of a stream (i.e. low gradient, low turbulence, moderately deep) then the contaminants have the potential to accumulate to significant levels on the riverbed in the immediate receiving environment.[6] In a higher energy stream environment, contaminants will be transported downstream. In this case there will be a tendency for them to be dispersed and therefore lower potential for build-up to ecologically significant levels in downstream depositional areas ('distant' receiving environments.)

In the case of estuarine and marine environments, there tends to be settlement of larger particulate material immediately adjacent to the discharge point and a wider dispersion of fine particulate matter as a result of tidal water movement or wave action. This can result in the formation of a 'halo' of larger particulate material and contaminant build-up in the sediment adjacent to inter-tidal and sub-tidal stormwater outfalls. The deposition of contaminants in these areas is augmented by the flocculation of aggregated fine particles at the freshwater-saltwater interface.

If the immediate receiving environment is strongly depositional (e.g. a deep, sheltered inner harbour subject to limited water movement, a low gradient stream, or a lake) a high proportion of the particulate load will be deposited in the immediate vicinity of the outfall.

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[6] Whether significant build-up of contaminant concentrations in sediments actually occurs depends in part on the natural sediment budget of the area, i.e. the extent of 'dilution' of contaminants by uncontaminated sediment deposited in the area.

In this event there will be high potential for the concentration of contaminants to build up in the sediments to ecologically significant levels.

If the receiving environment is *moderately depositional* (e.g. rivers of moderate gradient periodically flushed by flood-flows, or estuaries subject to moderate flushing by tide or wave-induced currents), then contaminants have less potential to accumulate and may be dispersed over a wider area. This can lead, for example, to contaminants discharged from roads to the upper parts of estuaries being gradually dispersed seawards; this effect may lower the potential for exceedance of sediment quality criteria in the medium term but perhaps increase the potential for significant widespread contamination in the longer term (see Auckland estuaries, Section 3.3 below).

If the immediate receiving environment is *high energy/dispersive* (e.g. a shallow fast-flowing river or an open coastline), particle-associated contaminants will tend to be rapidly dispersed and diluted. The particles will eventually settle in a distant receiving environment but the degree of dispersion that has occurred means that there will be limited potential for contaminant build-up in sediments.

B3 The effects of road runoff on receiving environments

B3.1 Introduction

Having considered the likely fate of the contaminants in road runoff on theoretical grounds (chemical state and receiving environment hydro-dynamics), it is important to consider scientific findings pertaining to the actual fate and environmental effects of road runoff on different types of receiving environment. (The latter has assisted with development of this methodology for the identification and ranking of sensitive receiving environments).

There is extensive literature relating to the effects of road runoff on receiving environments. Whilst laboratory-based testing has provided some useful information on the toxicity of specific contaminants, it is apparent, from New Zealand and overseas field studies, that the actual environmental effects of road runoff are site-specific. The effects depend on the nature of the contaminant load (e.g. specific contaminants, chemical state and total load), and a host of receiving environment variables including water movement; available dilution and dispersal; existing water and sediment quality (e.g. contaminant levels, pH, water hardness, dissolved organic matter content); ecological values and human uses.

A wide-ranging review of the environmental effects of road runoff is outside the scope of this research project. However, Kennedy (2003) has published the findings of such a review entitled “The Effects of Road Transport on Freshwater and Marine Ecosystems”. The report reviews both the overseas and New Zealand literature pertaining to the effects of urban and road runoff on aquatic ecosystems.

The salient findings from this paper have been determined during the project and are summarised below under the heading "Key MoT Findings".

B3.2 Effects of urban and road runoff on freshwater receiving environments

Key MoT findings are as follows:

- It is difficult to isolate the effects of contaminant runoff from roads and highways on streams from the effects of contaminants derived from other sources of urban runoff[7] such as industrial sites, residential properties and roof runoff.

- In New Zealand only a small number of studies have attempted to isolate the effects of runoff from roads or highways from the effects of general urban runoff.

- A number of overseas and New Zealand toxicity studies have shown that urban stormwater is capable of having both acute and chronic effects on freshwater organisms, fish and invertebrate species.

- Stormwater quality data from urban stormwater discharges and from road and motorway discharges in New Zealand shows that dissolved concentrations of copper and zinc can reach very high levels in the 'first flush' of stormwater.[8] The concentration of zinc is the most significant in relation to water quality guideline criteria such as ANZECC (2000). The regular exceedance of 'trigger' criteria for dissolved concentrations of zinc and copper in the water column suggests the likelihood of adverse effects on aquatic biota (Kennedy 2003).

- The toxic agent in some US urban stormwater studies has been identified to be organic compounds (in some cases pesticides). However, there appears to be little published evidence of organic compounds in New Zealand road runoff being toxic and causing adverse effects on freshwater biological communities.

- Based on the composition of suspended sediment, it is likely that concentrations of contaminants can build up in stream sediments adjacent to road runoff discharge points to levels that could have an adverse effect on freshwater benthic organisms.

- Depending on the ratio of contaminated stormwater sediment to uncontaminated sediment, elements such as zinc may exceed sediment quality

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[7] It is, however, possible to do and several overseas studies have been conducted into the effects of runoff from bridges, sections of highway isolated from adjacent land uses, parking areas and remote rural roads.

[8] Timperley 2002 provides data from the Otahuhu Motorway indicating that dissolved zinc and copper concentrations in runoff are in the order of 21 mg/m^3 and 70 mg/m^3 as against ANZECC (2000) triggers for 95% protection of 1.5 mg/m^3 and 8.0 mg/m^3, respectively.

guidelines at which adverse effects could be expected to occur on benthic organisms.[9]

- Contaminant concentrations in lake sediments adjacent to stormwater discharge from roads can approach or exceed sediment quality guidelines. For example, sediments collected adjacent to a road outfall in Hamilton Lake revealed elevated concentrations of lead and zinc that exceeded ANZECC guideline criteria (Rajendram 1992 in Snelder and Trueman 1995).

- Elevated concentrations of dissolved or particulate metals in road runoff have the potential to result in bio-accumulation in freshwater biota. Elevated concentrations of Cu, Pb, and Zn have been reported in a range of species but there is little or no evidence of adverse effects arising from such accumulation.

- There is a wide range of organic compounds emitted by vehicles that have the potential to bio-accumulate in aquatic organisms. The key groups are PAHs and substituted PAHs. Although uptake of these contaminants has been shown to occur, no adverse effects of bio-accumulation appear to have been identified.

- There is little information to indicate that any freshwater species (e.g. eel, watercress) exposed to metals or PAHs derived from road runoff would bio-accumulate these contaminants to a point that would result in them being unsuitable for human consumption from a health risk perspective.

B3.3 Effects of urban and road runoff on marine receiving environments

Key MoT findings are as follows:

- As with freshwater receiving environments, it can be difficult to isolate the effects of road runoff from the effects of contaminants from other urban sources[10].

- Most studies carried out to date in New Zealand have focused on the possible effects of urban runoff on coastal ecosystems.

- There appears to be very few, if any, studies of the level of dissolved contaminants in marine waters in the vicinity of stormwater discharges, possibly because of the substantial dilution available in the sea. (Note: this is an interpolated finding based on the lack of coverage of this issue in the MoT reports).

- Studies in New Zealand have shown that there is a build-up of contaminants adjacent to some urban stormwater outfall discharges to the marine environment

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[9] In urban areas, streams may not receive their normal supply of sediments and, as a result, sediments derived from roadways and other urbanised sources may dominate the stream.

[10] Namely, when road runoff is channelled into piped or open-channel urban stormwater drainage system as is common in urban areas.

and in inter-tidal areas adjacent to roads and highways (particularly in the upper parts of estuaries to levels approaching or exceeding ANZECC sediment quality guidelines).

- In low energy/depositional environments, such as inner harbours and estuaries, there is typically a 'halo' of sediment modification and contamination surrounding the outfall. The size of the halo is dependent on the magnitude of the discharge, its contaminant load and the hydrodynamic (water movement) characteristics of the receiving environment.[11] Levels of contamination adjacent to the outfall can approach or exceed sediment quality 'trigger' levels, that is, levels potentially dangerous to benthic organisms.

- In ecological terms, the most apparent effect of the discharges, within the halo, is the replacement of normal benthic communities with an impoverished fauna of hardy or opportunistic species. [Note – the precise nature of the cause-effect relationship is often not clear; changes in faunal composition can potentially be caused by changes to the physical composition of the sediments or to their chemistry, (e.g. ammonia or sulphide build-up as a result of organic material in runoff[12]), as well as by the acute or chronic toxic effects of vehicle-derived contaminants.]

- Notwithstanding exceedence of sediment quality trigger levels, there is no direct evidence of sediment-related toxicity nor that elevated contaminant levels are a causal agent for changes in the benthic community.

- In the wider area of urban harbours and estuaries, sediments often contain elevated concentrations of metal and organic contaminants as a result of dispersion from point-source discharges (e.g. in locations such as Waitemata Harbour it appears that all of the surface sediments are contaminated to some degree).

- In relation to the potential for bio-accumulation, it is unlikely that road transport contributes to any bio-uptake of contaminants by marine organisms, adjacent to roads and urban areas.

- Current knowledge provides no information to indicate that road transport contributes to the uptake of contaminants by marine organisms, such that they are rendered unfit for human consumption. Two of the key trace element contaminants – copper and zinc – are typically not bio-accumulated by many marine organisms.[13]

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[11] The halo is normally manifested as a 'bulls-eye' of diminishing contaminant concentrations moving away from the outfall. The area of discernible contamination can have a diameter of 100 m or more.

[12] Urban stormwater discharges typically contain a large amount of organic debris – leaves, paper, cardboard, cigarette butts, etc – from street runoff.

[13] Some shellfish (e.g. oysters) can accumulate zinc.

B3.4 Summary

In summary, it is apparent that:

- There is scientific evidence that the levels of dissolved metals exceed guideline limits for streams in the ‘first flush’ of urban stormwater runoff (and probably in the first flush of road runoff), but no direct evidence of adverse effects on stream organisms.
- There appears to be little evidence of build-up of road runoff–derived contaminants in the sediments of streams to levels representing a threat to benthic organisms, but it is likely that this does occur adjacent to some road runoff outfalls.
- There is evidence of build-up of contaminants adjacent to outfalls in marine depositional environments (i.e. sheltered inner harbours and estuaries) to levels approaching or exceeding sediment quality guidelines.
- There are typically changes to the composition of benthic fauna within the contamination ‘halo’ surrounding these marine outfalls, but direct causal relationships have yet to be established.

The primary concern, from the scientific and road management perspectives, is that if nothing is done to reduce road runoff contaminant inputs, contaminants could build up in sediments to levels that have serious ecological effects.

B4 Conclusions on key attributes for the sensitivity rating system

As outlined in Section 2.2, contaminants can be transported in road runoff in either the dissolved or the particulate phase, and it is the dissolved (soluble) phase that is capable of exerting an immediate toxic effect on aquatic life.

In the UK, studies carried out on the polluting effects of road runoff have focused on the dissolved fraction of contaminants and on riverine receiving environments, presumably on the basis that there are many slow-flowing low gradient lowland streams in the UK with limited dilution and dispersal, and coastal environments tend to have high available dilution. Another reason for this focus could be that the UK has an extensive, heavily-trafficked network of highways and trunk (main) roads which already have treatment devices aimed at minimising the runoff of solids.

In the case of dissolved contaminants, the principal factors determining the sensitivity of a receiving environment to adverse effects are the available dilution and the quality of the receiving water. In a UK study entitled “Identification of Outfalls Posing a Pollution Risk” (Highways Agency 2000), the primary inputs to the model that was adopted for assessing potential impacts on receiving watercourses were low flow and upstream water quality data.

In New Zealand, streams are generally of higher gradient, water velocity is greater, and hence there tends to be a higher assimilative capacity for dissolved contaminants.[14] This may explain why there has been relatively little concern about the effects of the dissolved contaminant component in road runoff in New Zealand. Although the dissolved fraction can build up to high levels in urban streams, possibly in exceedance of ANZECC trigger levels, in the 'first flush' of runoff (see Section 3.2) high levels of dissolved contaminants do not persist in the runoff.

The main focus of concern in New Zealand (amongst resource managers and researchers) has been the potential for build-up of contaminants in sediments adjacent to urban stormwater outfalls and road runoff outfalls, with a particular focus on sheltered marine receiving environments. The focus on the latter, rather than riverine receiving environments, probably reflects a number of factors including the settlement and road patterns in New Zealand and the dispersive nature of many New Zealand rivers and streams.

In some urban areas, particularly Auckland, considerable management effort is being directed to controlling inputs of particulate material into freshwater and marine receiving environments. This approach is justifiable because, as outlined in previous sections of this paper, the available scientific evidence indicates that:

1. A high proportion of the contaminants of concern (viz. metals, PAHs, and oils) is strongly associated with the solids fraction in road runoff.
2. There is a tendency for dissolved contaminants to absorb to particles during passage through the drainage system to the final receiving environment.
3. Contaminants associated with particulates demonstrably accumulate in certain depositional environments to levels exceeding sediment quality guidelines, e.g. in inner harbours and the upper parts of estuaries.
4. There is potential for uptake of contaminants from sediments by benthic organisms (and food chain accumulation) by way of incidental ingestion with food particles or by way of gill uptake of dissolved contaminants released from sediments into interstitial waters or the waters above sediments.
5. In contrast to the situation with dissolved contaminants in rivers, benthic organisms in sedimentary deposits can have long-term exposure to high contaminant levels.

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[14] It is appreciated by the research team that this is a generalisation and does not always hold true.

6. Particulates and associated contaminants have the potential to be deposited on ‘biofilms’ on rocks or macro-algae in rivers and in the marine environment, and incidentally ingested by grazing macro-invertebrates.

7. Particles from urban or road runoff can have adverse ecological effects in their own right by smothering biota or by altering the physical nature of sediments in the vicinity of outfalls.

8. The particulate fraction of road runoff is more amenable to removal (via source control, in-pipe and end-of-pipe treatment systems) than is the dissolved/soluble fraction.

The above considerations suggest that the focus in this research project on the particulate fraction of the contaminant load is appropriate and that the primary factor determining the impact of sediment-associated contaminants is whether or not the immediate receiving environment is depositional or dispersive. Solids will settle rapidly in still or slow flowing water and will tend to accumulate near the outfall. In more dispersive environments there will be limited potential for contaminants to accumulate.

This approach has guided development of the proposed methodology for ranking the sensitivity of receiving environments to road runoff, as described in Section 3.5 and Appendix C.

B5 References

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Appendix C: Rating the sensitivity of receiving environments to road runoff

C1 Rating framework

A preliminary framework for the sensitivity rating system of receiving environments to road runoff is given in Table C1.1. Receiving environments are sequentially classified according to their:

- Physical 'type' (depositional versus dispersive),
- Ecological values, and
- Human uses and values (including cultural values).

A description of these three attributes is provided in the following sections.

C2 Receiving environment type classification

The receiving environment type classification is regarded as the primary classification and weighted accordingly. This approach is based on the premise that low energy depositional or 'sink' environments, with little water movement, are at greatest risk of build-up of contaminants in sediments to levels representing a risk to aquatic organisms or human use values.

Conversely, high energy dispersive receiving environments with significant water movement (e.g. rapidly flowing rivers or open coastlines) are at low risk of adverse effects as contaminants are rapidly mixed, diluted and dispersed. In short, dispersive receiving environments have a higher assimilative capacity than depositional receiving environments.

Receiving environments can be broadly partitioned into 'Strongly Depositional', 'Moderately Depositional' and 'Dispersive' reflecting their intrinsic sensitivity (see Table C1.1).

C2.1 Strongly depositional receiving environments

Enclosed sheltered harbours and embayments and estuaries tend to be strongly depositional and these receiving environments, when near urban areas and/or adjacent to high density traffic roads, are the ones most commonly identified as having elevated levels of contaminants in their sediments, particularly in the vicinity of stormwater outfalls.

Table C1.1 Framework for assessment of the sensitivity of receiving environments to road runoff.

A. Primary classification: receiving environment type.

Type of Receiving Environment	Sensitivity	Score	Information Sources
Strongly Depositional • Enclosed/sheltered harbour, embayment • Estuaries • Low gradient/velocity streams or rivers • Small lakes, some larger lakes • Wetlands	**H**	**30**	Visual observation Map Inspection GIS Databases
Moderately Depositional • Semi-enclosed harbours, embayments • Moderate gradient/velocity streams or rivers • Large lakes	**M**	**20**	
Dispersive • Open/exposed coastal environment • High gradient/velocity streams or rivers	**L**	**5**	

Table C1.1 (continued) Framework for assessment of the sensitivity of receiving environments to road runoff.

B. Secondary classification: ecological values.

Ecological Values	Sensitivity	Score	Information Sources
• Water body has high formal conservation status e.g. within a national park, reserve, marine reserve, wildlife refuge, protected natural area or identified as regionally or naturally significant, or • Rare, threatened, endangered species present (flora or fauna), or • Plant or animal community with high species diversity, or • Rare habitat or good representative example for region, or • Particularly valuable habitat eg whitebait spawning area, or • Particularly high quality habitat/water present e.g. upper reaches of some streams or springs.	**H**	**20**	Regional Councils Regional Plans Regional Registers or Databases Department of Conservation NZ Fish and Game Environmental Groups Iwi
• No formal conservation status, and • Absence of rare, threatened, endangered species, and • Moderate species diversity, and • Moderate habitat diversity, and • Habitat values moderate.	**M**	**10**	
• No formal conservation status, and • Absence of rare, threatened, endangered species, and • Low species diversity, and • Low habitat diversity, and • Habitat values low e.g. significant physical modification and/or contaminant inputs from sources other than road runoff.	**L**	**5**	Regional Councils Regional Plans Regional Registers or Databases Department of Conservation NZ Fish and Game Environmental Groups Iwi

Table C1.1 (continued) **Framework for assessment of the sensitivity of receiving environments to road runoff.**

C. Tertiary classification: human uses and values [(a)]

Uses/Values	Sensitivity	Score	Information Sources
• Highly used or valued for food gathering including traditional Maori food sources (e.g. shellfish, koura, watercress), or • High use for contact recreation (e.g. swimming, paddling), or • High use for non-contact recreation (e.g. fishing, canoeing, windsurfing, boating), or • High cultural/spiritual values associated with water body, or • Downstream water supply, or • High economic value e.g. tourism.	**H**	**10**	Regional Councils Iwi Recreational Groups
• Highest status reasonably assigned to the receiving water, in terms of all of the above uses or values is 'moderate' (see guidance in text).	**M**	**5**	
The status reasonably assigned to the receiving water, in terms of all of the above uses or values, is 'low' (see guidance in text).	**L**	**2**	

TOTAL SCORE		(minimum 20 and maximum 60)

Notes: Overall Sensitivity Rating:

high sensitivity = total score >40
medium sensitivity = total score 20-40
low sensitivity = total score <20

(a) Some users of the methodology may wish to remove aspects concerning Maori culture/values from the weighting table (and hence from assessment of human values) and treat this as a separate qualitative decision-making criterion

For example, trend analysis of the Auckland Regional Council's Long Term Baseline Monitoring Programme shows that zinc and copper concentrations in sediments in the upper parts of Auckland's urbanised estuaries are increasing. In strongly depositional environments, impacts are likely to be local as there is little or no remobilisation of solids beyond the initial point of settlement.

In the case of low gradient, moderately deep, streams and rivers (usually in the lower parts of catchments), water velocity is low and there is potential for the contaminants attached to fine-grained sediments to accumulate near the point of discharge. Depending on the ratio of contaminated particles to 'natural' uncontaminated sediment, there is potential for contaminant sediment to build up to levels harmful to aquatic life (see Section B3.2, Appendix B).

In some cases, such streams or sections of streams, may be subject to periodic flushing of the sediments by floodwaters. This is an issue that needs to be considered when deciding whether or not a receiving environment should be classified as strongly depositional. (If the receiving environment is so flushed, it is probably better classified as 'moderately depositional' reflecting the lower degree of risk.)

Lakes (more or less static water bodies with no tidal or oceanic currents) tend in general to be strongly depositional environments due to the relative lack of water movement. Consequently, ecologically significant 'halos' of sediment contamination can build up adjacent to shoreline outfalls. However, some lakes, depending on their size and degree of exposure to winds can have significant wave action that can help to disperse contaminants and therefore make them less sensitive to the effects of runoff. Some larger lakes with shallow shorelines fall into this category and may therefore be best classified as 'moderately depositional'. Most small lakes are best classified as 'strongly depositional' due to reduced water movement and generally lower assimilative capacity.

Wetlands, which are generally static waterbodies with vegetation impeding the movement of water, are usually appropriately classified as strongly depositional.

C2.2 Moderately depositional receiving environments

Semi-enclosed harbours and embayments, with some water movement, should be classified as 'moderately depositional' even though there may be some doubt about the extent to which contaminants are likely to accumulate in bottom sediments.

Similarly, rivers and streams of moderate gradient and water velocity should be classified as moderately depositional. As indicated above, there may be a case for classifying some low gradient streams as such on the basis of considerations such as water depth and likelihood of sediment flushing during floods.

Some larger lakes may be classified as moderately depositional if there is strong wave action and shallow water adjacent to the subject outfall.

C2.3 Dispersive receiving environments

Open, exposed, high energy coastal environments should be classified as ‘dispersive’ as contaminants in road runoff – be they dissolved or attached to particulates – will be rapidly dispersed away from the discharge point, particularly where the discharge is to the intertidal area or above the wave base which is commonly the case.

High gradient, fast flowing rivers, particularly shallow rivers (or streams), are turbulent and both dissolved and particulate-associated contaminants will be rapidly diluted and dispersed downstream. In other words, they do not accumulate in the immediate receiving environment.

However, fast-flowing rivers effectively act as conduits for the pollutant load that accumulates downstream such as an estuary or harbour. While such rivers should be classified as ‘dispersive’ receiving environments, the assessment will need to consider impacts on the distant depositional receiving environment.

C3 Ecological values classification

The ecological values associated with a receiving environment are clearly an important consideration. Assignment of a high priority to ecological criteria in assessing receiving environment sensitivity is consistent with the Resource Management Act’s emphasis on avoiding adverse effects on ecosystems, safe-guarding life support systems, protecting significant vegetation and habitats of indigenous fauna, and sustaining the potential of natural resources to meet the reasonably foreseeable needs of future generations.

The methodology treats ecological values as a secondary consideration relative to receiving environment type and the scores are weighted accordingly. This is because if the latter is dispersive there is very little risk to the receiving environment (i.e. it is relatively ‘insensitive’), irrespective of the ecological values present.

The proposed ecological criteria (values) shown in Table C1.1 are preliminary and the way they have been classified into groups of ‘high’, ‘medium’ and ‘low’ ecological status are open to debate and could be reviewed in discussion with relevant agencies and interested parties. However they are considered appropriate for use in the sensitivity rating tool developed under this project (see Section 3.5 of main report).

Information pertaining to ecological criteria would need to be sought from relevant agencies, including regional councils, Department of Conservation, and NZ Fish & Game. In some cases, information is likely to be readily available from regional councils in the form of existing registers, plans or GIS databases.

C4 Human uses and values classification

The human uses and values associated with a given receiving environment are an important consideration.

The objective of protecting significant human uses and values associated with water bodies is consistent with RMA requirements to avoid, remedy or mitigate adverse effects on the environment (widely defined to include people and communities); to recognise and provide for the relationship of Maori and their cultural traditions with their ancestral lands, water, waahi tapu and other taonga; and to have particular regard to the maintenance of amenity values and the quality of the environment. (Note: It is acknowledged that some users of the methodology may wish to remove aspects concerning Maori culture/values from the weighting table, and hence from assessment of human values, and treat this as a separate qualitative decision-making criterion.)

Notwithstanding the above, the proposed methodology for assessing and ranking the sensitivity of a receiving environment treats this as a tertiary consideration (reflected in weighting of scores). The reason for this is that the RMA places emphasis on safeguarding life support systems and protecting ecosystems (so that there *are* options for future generations and because many human uses and values are fundamentally dependent upon the maintenance of ecological values).

C5 Disturbed ecosystems

An issue that needs to be considered, in some situations, is the extent of contamination (water quality or sediment quality) or disturbance that is already present and how this affects the sensitivity of a receiving environment (waterbody).

As indicated in Appendix A, a disturbed ecosystems can be viewed in two quite different ways – that the environment is already degraded and therefore not so sensitive, or that it is more sensitive because maintaining or increasing the contaminant load could increase the stress on the ecosystem, either pushing it over an ecological ‘break point’ or increasing the size of the impact zone.

This issue can be relevant to the decision as to whether or not to install a runoff treatment system, either retrospectively or when constructing a new road. It may also be relevant during stages that might avoid SREs, such as during route selection.

In the UK study (UK Highways Agency, 2000) aimed at identifying road outfalls posing a pollution risk, the approach taken was that lower quality receiving waters (rivers), having been identified, were eliminated from further consideration. The rationale for this approach was that such water bodies were considered to be relatively insensitive to road runoff, therefore at low risk and not warranting the installation of new or additional treatment devices to protect the receiving water.

In Australia, the Road Transport Authority procedures for selecting treatment strategies for road runoff provide no qualification on how to rank water bodies with existing levels of contaminants or disturbance. However, the ANZECC (2000) water quality guidelines recognise three ecosystems 'conditions', viz.:

i) Effectively unmodified systems (high ecological value).

ii) Slightly-to-moderately disturbed systems: ecosystems in which aquatic biological diversity may have been adversely affected to a relatively small but measurable degree by human activity. The biological communities remain in a healthy condition and ecological integrity is largely retained. These ecosystems could include rural streams receiving runoff from farmland, or marine ecosystems lying immediately adjacent to metropolitan areas.

iii) Highly disturbed systems: these are measurably degraded ecosystems of lower ecological value. Examples of highly disturbed systems would be some shipping ports and sections of harbours serving coastal cities, urban streams receiving road and stormwater runoff, or rural streams receiving runoff from intensive horticulture.

The third ecosystem condition recognises that degraded aquatic ecosystems still retain, or after rehabilitation may have, ecological conservation values but for practical reasons it may not be feasible to return them to a slightly-to-moderately disturbed condition.

ANZECC (2000) recommends that these levels of ecosystem condition be used as a framework to decide upon an appropriate level of protection. Key stakeholders in a region would normally be expected to decide upon an appropriate level of protection through determination of management goals based on the community's long-term desires for the ecosystem. This decision would consider matters such as the assimilative capacity of the water body, the ability to 'clean up' existing contamination and/or reduce existing contaminant load or other disturbances, and the ability to recover if stress levels are reduced. The ANZECC philosophy behind selecting a level of protection, inherent in the water quality guidelines, is either (i) maintain the existing ecosystem condition, or (ii) enhance a modified ecosystem by targeting the most appropriate condition level.

As indicated in Section A4.3 of Appendix A, the Hawke's Bay Regional Council's draft methodology for assessing the sensitivity of receiving environments identifies that the degree of existing degradation has a bearing on the (remaining) assimilative capacity of a receiving environment and the potential for cumulative effects. It proposes a receiving environment classification system that takes into account the existing degree of degradation as a tertiary consideration.

Receiving environments will often not be significantly degraded or disturbed and the community will often not have given consideration to the issue of (realistic) management objectives for a given water body. For this reason, the research team considers that the issue of existing degree of contamination/disturbance is best left out of the proposed 3-step classification of receiving environment sensitivity described above. Moreover, the best approach to dealing with this issue is for all parties to recognise that there will be some situations where existing or 'baseline' disturbance is relevant to decisions about whether or not new or additional treatment measures to control road runoff should be installed.

In such cases, the regional council should be asked to advise on whether or not a (community) consensus has been reached on the appropriate degree of protection of the receiving environment in question as per the ANZECC recommendations discussed above. If such agreement has not been reached, an important issue will be the likely contribution of road runoff to the total contaminant load on the receiving environment in question. If it is low, there may be a weaker case for the installation of road runoff treatment systems but this is a matter that should be the subject of consultation with stakeholders.

C6 Groundwater receiving environments

Stormwater discharging into a roadside drain, grass swale, rain garden or other form of ground soakage may seep directly into the shallow groundwater. Its passage through surface soils tends to filter out particulate material, which is retained at on near the surface of the soil profile. Stormwater entering the groundwater can therefore be expected to be substantially free of particulate material but may not necessarily have reduced concentrations of dissolved contaminants. (Retention of dissolved contaminants would depend, amongst other factors, on properties of the soil matrix).

In the UK, there is no evidence currently available to suggest that the contamination of groundwater from routine road runoff is a serious problem (UK Highways Agency 2000). However, the risk of road runoff contaminating groundwater resources (particularly in the event of a spill[15]) is taken seriously and regulatory authorities do not accept point source discharges to the ground in areas that are susceptible to groundwater pollution (denoted Source Protection Zones). In the source catchment, road discharges to ground are only permitted if the results of an investigation are favourable and if adequate precautions are taken such as installation of oil separators (particularly relevant in the event of a spillage) and a properly designed soakaway aimed at removing the solid phase.

In New Zealand, with lower road and traffic densities, the risks to groundwater resources from road runoff are lower than in the UK (where no significant problems appear to exist). Consideration of groundwater as a receiving environment has therefore been excluded from the scope of this research project, which has a focus on the particulate fraction and depositional environments at risk from road runoff.

This is not to say that relevant risks, such as those from verge-side chemical use or tanker spills, should not be recognised and taken into account when considering new road routes and treatment system issues. Some district plans identify aquifer recharge or

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[15] Spillage issues and risks have not been specifically considered as part of this research project but the SRE assessment methodology developed herein could be applied to consideration of spillage risks and development of response protocols.

protection zones that should, if practicable, be avoided when planning a new route. Risks to groundwater arising from roadside herbicide and pesticide use can only be effectively reduced through judicious application and selection of the least persistent chemicals.

C7 References

ANZECC, 2000. Australian and New Zealand Guidelines for Fresh and Marine Water Quality. Australian and New Zealand Environment and Conservation Council.

UK Highways Agency, 2000. Identification of Outfalls Posing a Pollution Risk. *Report prepared for UKHA by Thorburn Colquhoun, October 2000.*

Appendix D: Literature review of factors affecting quality of road runoff

D1 Introduction

The primary aim of the literature review was to identify the main source risk factors (i.e. within the road environment) that contribute to or affect the quantity of contaminants in road runoff – and therefore place downstream SREs potentially at risk. These are the factors that influence the quality of road runoff, as measured in terms of contaminant load, or concentration of contaminants and volume of runoff.

The main vehicle-derived contaminants of concern in road runoff are heavy metals (notably copper and zinc) and PAH (see Section 2 of Appendix B).

The following section reviews traffic, road and rainfall factors as these relate to runoff quality. This is followed by a discussion of different approaches for modelling pollutant loads in road runoff, and related Ministry of Transport research. The section is concluded with a review of screening methodology used for assessing the impact of traffic on the aquatic environment.

D2 Factors affecting road runoff quality

The factors affecting quality of road runoff are numerous and complex with many of these being interrelated. Contaminants on roads may have other sources apart from vehicles e.g. contributions from outside the road environment such as zinc from roof runoff. Furthermore, a fraction of particulate contaminants that build up on road surfaces may subsequently be dispersed by winds before they become washed out to the road drainage system. In urban areas, the process of street cleaning will also reduce the contaminant load in road runoff.

The main factors affecting quality of road runoff can be grouped broadly under the following headings:

- Traffic conditions and road characteristics,
- Rainfall and runoff patterns, and
- Road drainage infrastructure.

The first two factors are concerned with contaminant accumulation on the road surface. The third factor affects the contaminant wash-off process. The type of road drainage (e.g. kerb and channel, catchpit, swale) plays a key role in controlling the pollutant load leaving a road reserve, and therefore the risk to downstream receptors.

D2.1 Effects of traffic and road characteristics

The primary risk factor to road runoff is the density of traffic as this is the main driver of vehicle-derived contaminants. For this reason it is expected that a direct correlation should exist between traffic levels and the quality of road runoff.

While broad relationships have been identified from field measurements between contaminant levels in road runoff and average vehicle flows, the literature indicates that pollution from road runoff is very variable in nature and the relationship between traffic density and runoff quality is complex. For example, an Australian study (Drapper et al. 2000) found that traffic density and runoff quality from highways were only weakly correlated and not sufficiently robust to propose traffic volume as the best indicator for roads requiring runoff treatment.

More recent research has shed light on the influence of related variables such as road characteristics (e.g. hills, bends, intersections, traffic lights etc) that can affect driving conditions and traffic congestion and, as a result, influence vehicle emission factors under local conditions. Thus, contaminants derived from brake pad wear (e.g. copper) or tyre wear (e.g. zinc) will be influenced by traffic and road attributes affecting braking such as degree of traffic congestion, presence of intersections and topography (e.g. hills/bends).

Roads with the same traffic flows but markedly different road characteristics are found to generate very different contaminant loads in runoff. A similar effect is found for roads of similar type but with different levels of traffic congestion (see examples in Appendix E, Section E4.6).

Muschack (1990) found that driving patterns influence contaminant loads. Traffic lights on a section of highway compared to one without increased the levels of pollutants. This was attributed to an increase in braking and acceleration with a concomitant increase in brake lining wear, tyre abrasion and leakage of oil and exhaust gas emissions.

Drapper et al. (2000) identified elevated levels of heavy metals in road runoff from exit lanes on a bridge. The effect of rapid vehicle deceleration at the exit resulted in increased brake and tyre wear and higher concentrations of copper and zinc, compared with sites without an exit lane.

The effects of varying road and traffic characteristics on the emission factors of contaminants from vehicles is discussed more fully in the New Zealand context by Kennedy et al. (2002).

In summary, reported variations in the quality of road runoff are partially due to differing pollutant loads, attributable to differences in traffic/road conditions. Therefore, a model to prioritise road sources in terms of contaminant load in runoff should take the factors influencing these differences into account.

D2.2 Rainfall and runoff patterns

Rainfall is the medium for removing pollutants that have built up on the road surface and is therefore a key aspect affecting the concentration of contaminants in road runoff.

The relationship of rainfall to surface water quality has been extensively studied. The relationship is complex and a function not only of the amount of rainfall but other related variables such as the intensity and duration of rainfall during storm events, and the number of preceding dry days.

O'Riley (2002) cites a study by Stotz (1987) which showed that the concentration of contaminants discharged in surface runoff was not directly dependent on traffic density. Instead it was found to vary with wind conditions, rainfall frequency and intensity, runoff volume and the number of preceding dry days. This was confirmed by Ball et al. (1998) who suggested that average daily traffic (ADT) is significant only on a broad scale, with site-by-site variations related to other factors, predominantly the number of preceding dry days. Ball concluded that the relationship between the quality of road runoff and traffic density required further clarification.

The concept of `first flush' is that a large fraction of the contaminant material accumulating on an impervious road surface is flushed off into the drainage network during the early stage of a storm. Overseas research has shown that the peak contaminant concentrations during first flush can vary widely during an individual event (Sansalone and Buchberger 1997) and between events (Marsalek et al. 1997). For this reason the concentration is generally presented in terms of the event mean concentration.

The occurrence of a first flush is mainly dependent on the long-term build up of materials on the road surface and the magnitude of the rainfall event (Lee et al. 2002). The first flush may also be influenced by material from previous storm events that are flushed from road drains e.g. catchpits or stormwater pipes (Kennedy 2003).

O'Riley et al. (2002) studied a suburban roundabout carrying moderate traffic levels and confirmed that the first flush effect occurs under New Zealand road conditions and is similar to that reported overseas. The first flush was demonstrated for particulate-bound metals and PAH, however the pollutant wash-off patterns for dissolved metals were reported to be variable and inconsistent.

More recent research conducted in New Zealand to estimate the contribution of road runoff to total stormwater loads mass loads (Timperley et al. 2003; Timperley et al. 2005) has involved modelling contaminant accumulation and wash-off processes. These studies have demonstrated clearer relationships between the accumulation of dissolved and particulate metal concentrations in road runoff and individual rain events.

The significance of first flush has been used to advantage by designing stormwater drainage systems that capture the initial runoff, and thus are able to remove the fraction of discharge containing the highest concentrations of contaminants (Barrett et al. 1998).

Rainfall and runoff patterns are key risk factors in the quality of runoff from individual storm events. However, in the context of this project, and as discussed in the Stage 2 report, environmental impacts from road runoff are more aligned to the cumulative build-up of contaminants in sediments within the receiving waterbody. Over the long term, therefore, they are relatively insensitive to whether the pollutants that build up on a road surface are washed off in a single large storm event or a series of smaller events. In other words, the key determinant is the total pollutant load received by the waterbody in the longer term.

While the factors that determine the build-up of contaminants on a road surface and their release to stormwater are complex, in the long term it may be assumed that all such contaminants are washed off into runoff. On this basis, for a given load to the road surface from vehicle-derived pollutants, the pollutant load discharged in runoff is independent of rainfall intensity or quantity.

Accordingly, for the comparison and ranking of roads that may adversely affect sensitive receiving environments, it is appropriate only to consider the relative pollutant load corresponding to the estimated quantity of contaminant in runoff, independent of the contaminant concentration, rainfall or runoff pattern.

D2.3 Road drainage infrastructure

Highways with kerb and channel generate higher volumes of stormwater runoff and pollutant loads. A significant proportion of rural state highways in New Zealand do not have kerb and channel with the result that the pollutant load entering nearby water bodies is reduced. The benefit of using swales alongside highway verges to further reduce downstream pollutant loads is well known.

Road drainage infrastructure is considered in more detail in Appendix E, Section E4.4 as an integral part of the contaminant load model.

D3 Modelling road runoff pollution

D3.1 Modelling approach

Internationally, modelling of road runoff contamination can be divided into two categories:

i) Empirical models
ii) Source- based models

Empirical models

Empirical models are based on in situ measurement of runoff at sites with well-defined road/traffic and landuse characteristics, and derivation of an empirical relationship between the measured variables.

A large number of models have been developed to assess the loads of contaminants derived from stormwater runoff. The most comprehensive model in common use is the USEPA Stormwater Management Model (SWMM) which can cover single event and continuous simulation of urban stormwater runoff quality and quantity.

Several authors have reported on the estimation of pollutant loads from roads, highways and urban areas (e.g. Legret & Pagotto 1999). In these cases, the loads in stormwater runoff are estimated from the event mean concentrations (EMC) and the volume of the event. Total loads are then estimated by summing the loads per rainfall event for the defined catchment.

Other studies have used measurements at selected road sites to derive an empirical relationship between pollutant load and traffic levels. This approach is exemplified in New Zealand by the work of Timperley et al. (2003, 2005).

Empirical models provide estimates of relative contaminant loading for differing traffic levels or landuses. This is an effective method of assessing broad scale impacts of land use modifications and stormwater treatment options. However, contamination load rates within these models generally ignore or otherwise do not adequately account for differing road types or traffic volumes. For this reason an empirical approach is not appropriate in the context of this study.

Source-based models

Source models are based on source contributions and estimates of emission factors derived from contaminant concentrations and emission rates. For example, for zinc as a vehicle-derived contaminant, the pollutant load to the road surface is derived from the sum of individual sources including contributions from tyre wear, oil leakage and exhaust emissions. Factors are included to account for the proportion of contaminant available for washout to the drainage system.

In New Zealand, this approach was followed in the Ministry of Transport's research programme (see below).

D3.2 Ministry of Transport research

The Ministry of Transport (MoT) identified effects on water quality and associated aquatic ecosystems as one of the key impact areas of road transport. MoT (1996) reported that the contribution of road run-off to the stormwater contaminant load was uncertain but was considered to account for 40-50% of metals in aquatic ecosystems. However, it was acknowledged that research in New Zealand at the time was not sufficient advanced to quantify the contribution made by road transport to any specific effects.

For this reason, the MoT initiated a research programme to examine the effects of road transport contaminants on aquatic ecosystems. A series of research reports was prepared between 2000 and 2005 by Kingett Mitchell Limited, in association with Fuels and Energy Ltd, on behalf of the Ministry (see MoT 2006 for a full listing). The reports aimed to provide an overview as to what is known about the effects of the contaminants released by motor vehicles in a New Zealand context. Moncrieff & Kennedy (2004) provide a summary of the background and context of this programme.

Building on the model development research for vehicle emissions to air, a programme for developing a Vehicle Fleet Emissions Model for water (VFEM-W) was initiated. An overview of VFEM-W is described in Moncrieff & Kennedy (2004) and a flowchart of the process is included in Figure 8.2 of their paper. Extensive research was conducted to characterise pollutant loads from vehicle and road interactions. The VFEM-W model was intended ultimately to link source, pathway and ecosystem effects, on a geospatial basis, to assist evaluating options on stormwater management.

VFEM-W was designed to provide estimates of pollutant loads from road runoff based on derived emissions factors from five key vehicle-derived sources (brake wear, tyre wear, oil leakage, exhaust emissions and road surface wear). The design included provision for modifying these emission factors based on varying driving conditions and for linkage to the New Zealand vehicle fleet profiles. The model outputs (fleet-weighted average emission rates of individual contaminants, by vehicle type, road type and driving condition) were to be integrated with the local traffic network model to calculate contaminant loads for the road network.

Initial validation and calibration of the prototype VFEM-W model is described in Kennedy & Gadd (2003). This comprised field measurements of dust from road surfaces and gutters on roads in Waitakere City and a comparison of emission factors (mg/VKT) derived from these field studies with emission factors assumed by the model for road and traffic characteristics. Further details are discussed in Section D5 of this Appendix.

The contaminant emission factors that form the basis for the MoT's VFEW-W model are described by Kennedy et al. (2002). However, the equations relating emission factors and road/traffic conditions to contaminant loads have not been published and output from the MoT's research programme did not include an operational version of the VFEM-W model for public use.

For the purposes of this project, the extensive research published by the MoT has been used as a basis to derive a simple vehicle contaminant load model for estimating pollutant load in runoff.

D4 Screening criteria for traffic impact assessment on waterbodies

A number of studies have developed screening tools for impact assessment of road networks on receiving waterbodies. These have been used, for example, to assist identifying road networks – or sections of roads with specific outfalls - that may potentially have an adverse effect on the receiving environment. These studies are briefly reviewed below with a comment on their application to the current project.

D4.1 AADT as a traffic threshold screening indicator

While AADT is not in itself a robust indicator of road runoff quality, this measure of traffic has been used overseas as a screening tool to filter out roads that are unlikely to have a demonstrable effect on receiving environments. This section briefly reviews overseas practice and draws conclusions for the current research project.

UK CIRIA Stage 1 assessment

The CIRIA Stage 1 assessment (CIRIA 1994) has for many years been used a preliminary screening tool to determine whether routine road runoff poses a risk to surface waters. For metals (specifically copper and zinc), the impact is assessed in terms of the dissolved component and the extent to which this is diluted by the receiving water (dilution factor is the ratio of runoff volume to the 5 percentile river flow).

A matrix relating average traffic flow (AADT) to available dilution enables a decision to be taken on the need for further assessment of pollutant abatement using the Stage 2 method. For example, for traffic flows between 5,000-15,000 (AADT), no further risk assessment is required where the dilution factor is 3 or greater. For the next traffic tier (15,000 to 30,000 AADT), the threshold rises to 4 or greater.

In terms of an effects threshold for traffic levels, the UK Highway Agency guidance for assessing the effects of road runoff (DMRB 1998) notes that several studies have identified virtually no noticeable effects of drainage on receiving water quality at an AADT of less than 15,000 vehicles per day (VPD). In the AADT range 15,000-30,000, only minor impacts have been reported.

In the UK, traffic flows vary from single carriageways - carrying in the order of 10,000 VPD - to dual 4-lane motorways carrying up to 150,000 VPD. In New Zealand, typical AADT levels on state highways are lower e.g. a median of approximately 30,000 VPD compared with more heavily trafficked overseas highways.

Although it is difficult to generalise, traffic levels in excess of between 5000 to 10,000 AADT would be required before the generation of adverse effects from direct road runoff on sensitive receiving environments in New Zealand (M. Timperley, *pers comm*.).

UK Highways Agency - risk assessment of outfalls

The UK Highways Agency (2000) described an approach to identify outfalls on main roads and motorways that are likely to cause a level of pollution that require the fitting of new or additional treatment measures. The project was a comparative study of the outfalls rather than looking at the absolute pollution threat. The `high risk' outfalls were ranked according to the perceived risk posed to surface watercourses. The main problem areas identified by the study occurred where the busiest and older roads (i.e. those lacking pollution abatement in the design) crossed areas of low-flow high quality rivers.

The methodology used a filter to screen out certain roads on the basis of a combination of low traffic volumes and less sensitive surface water. Thus, roads with current traffic flows less than 15,000 AADT and which cross receiving waters of Class C or lower were excluded from the study. A second filter was then applied to the residual road network to highlight the `high risk' outfalls. The criteria were `all roads where current traffic flows exceed 30,000 AADT' or `all roads where the CIRIA Stage 1 assessment would fail' (see above).

Application of the UK Highway Agency AADT screening method would be difficult in New Zealand, as not all rivers have assigned environmental quality standards.

US Environmental Protection Agency practice

The US EPA (1996) acknowledged that, while the quantity of runoff generated depended on the frequency, intensity and duration of rain in the area, the quantity of pollutants originating from highway vehicles is not well understood. Nevertheless, research on highway runoff quality found that significant effects only occurred from highways (major freeways and urban arterials) with traffic volumes exceeding about 30,000 VPD. This value was the threshold used by the US EPA to determine whether runoff required treatment.

Implications for using AADT as a screening indicator in New Zealand

Both the CIRIA and Highway Agency screening methods are targeted at the dissolved metal component - there is no equivalent screening method for the insoluble (suspended) metal component. This was an issue highlighted by the UK Highway Agency's Design Manual (DMRB 1998) which states that "no simple method for determining the need for control of settleable materials in road discharges is currently available".

For this reason, these screening methods do not have direct application to the current project which is focussed on effects of settleable particles on depositional environments (see Appendix B "Development of an SRE sensitivity rating system" for a more detailed rationale).

A further limitation with AADT as a screening tool is that depositional environments are potentially affected by indirect as well as direct runoff contributions. While an AADT threshold may be set for a road section that discharges directly to a waterbody, this will take no account of the effects of indirect runoff that may be discharged from the same road further up the catchment. A further difficulty with the AADT threshold concept is that a series of road sections that discharge directly to an adjacent waterbody may each fall below the threshold but together exert an adverse cumulative effect on the waterbody.

D4.2 VKT as a traffic threshold screening indicator

An alternative traffic level screening indicator to AADT is total vehicle kilometres travelled (VKT). VKT is the product of AADT and road length, hence in a defined area is a measure of traffic density that is easy to depict spatially by GIS.

Brown and Afflum (2002) describe a GIS-based environmental modelling system using VKT for assessing the environmental effects of road transport plans. The system (known as TRAEMS – Transport Planning Add-on Environmental Modelling System) was developed using MapInfo and integrates land use information in the vicinity of the modelled road transport network.

The stormwater quality module of TRAEMS uses total VKT on roadways within a catchment (or sub-catchment) as a surrogate measure of pollutants that may affect waterbodies. The assumption is made that roadway emissions within a particular catchment will largely be washed off within that catchment. The model output is the relative pollution potential (RPP), indicative of the potential vehicle pollution load across sub-catchments.

An example application of the model in determining potential water pollution impacts based on road traffic growth forecasts for Gold Coast City is given in Brown et al. 2004. The authors acknowledge that the assumptions made in the model have not yet been validated.

The use of VKT at the sub-catchment level (and the RPP concept) appears to be a promising screening tool for identification of potential areas of road networks where runoff may have elevated contaminant loads that place an SRE at potential risk. It may be of value in the current project as the first stage (Tier 1) of screening a road network to identify potential `hot spots' that are subject to more detailed scrutiny under Tier 2 using a contaminant load model.

D5 Implications of findings

The literature review has considered the significance of risk factors to SREs at source i.e. within the road network before transport of runoff along the pathway to the receiving environment. The main findings are summarised below:

Factors affecting quality of road runoff

- The literature indicates that pollution from road runoff is very variable in nature and has a complex relationship with runoff quality. Traffic is the main source of road runoff pollution with the main contaminants of concern being heavy metals (notably copper and zinc) and PAH.

- Road characteristics and driving conditions have a strong local influence on vehicle emission rates, and hence contaminant loads. Models for estimating vehicle-derived contaminant loads in road runoff need to factor in these dependencies.

- For the current project, the contaminant load (mass) rather than contaminant concentration is the key information required from the perspective of determining the risk posed by traffic (the source component) to depositional environments. This requirement indicates the need for incorporation of a contaminant load model in the overall methodology for identifying SREs at risk from road runoff.

- An extensive and recent body of New Zealand research (published in 2001-2004) has been developed under the Ministry of Transport's programme on road transport and its effects on aquatic ecosystems. It is intended to use this research as the principal source of information on sources of contaminants and related vehicle emission factors applicable to New Zealand conditions.

Models for estimating contaminant load in road runoff

- Models for estimating road runoff contaminant load use emission factors for vehicle-derived pollutants as a function of traffic density e.g. expressed as mg/VKT. The contaminant load is calculated by multiplying the contaminant emission factor by the vehicle flow and the length of the road.

- Models for road runoff have been developed in New Zealand to provide estimates of contaminants (e.g. copper, zinc and lead) expressed in terms of mg/VKT. The models are either empirical (i.e. relationship based on field measurements of the variables) or source-based (i.e. relationship developed from assumptions on pollutant emission factors for different source components).

- Contamination load rates within empirical models generally ignore or otherwise do not adequately account for differing road types or traffic characteristics that have a defining influence on pollutant load. For this reason a source-based model is the preferred approach in the context of this study.

- The MoT's research programme (road transport impacts on aquatic environments to predict vehicle-derived pollutant loads in the environment from road runoff) included work to develop a source-based model (VFEM-W). A published study on validation of this model in Waitakere City showed that load predictions were weakly correlated to measured levels of copper and zinc from road surfaces. No working model was issued from this programme and the equations relating emission factors and road/traffic conditions to contaminant loads have not been published.

- Derivation of a working source-based model to estimate contaminant loads on a comparative rather than absolute basis, and based on the MoT's extensive published data, would be an appropriate way forward for the current project.

Screening tools

- Screening tools for impact assessment of road networks on receiving waterbodies have been briefly reviewed. Tools developed overseas typically incorporate AADT banding as a means to differentiate risks from roads and are generally based on effects on water quality at the discharge point rather than the impact of pollutant load on depositional receiving environments.

- Traffic levels measured as AADT are a poor proxy for runoff quality. A further limitation with AADT as a screening tool for effects on depositional environments is that it is primarily designed for direct impacts on a waterbody, and will not include indirect contributions that may occur higher up the catchment. It also does not take into account cumulative impacts from road sections that discharge to the same waterbody.

- A more robust screening indicator of source risk is total VKT in the catchment that drains to the SRE (rather than AADT on any given road section), in order to take account of indirect contributions and the cumulative risk from the road network. Published research indicates that VKT at the sub-catchment level is a promising approach for identification of road networks where runoff may potentially have elevated contaminant loads.

On the basis of the above findings, the following two-tiered approach is recommended for identifying and characterising the risk to SREs from traffic-sourced contaminants in road runoff:

i. A screening tool for identifying potential `hot spots' - areas of high traffic density that may have an adverse impact on water quality; this is to be based on total VKT by subcatchment and to be developed for both state highways and local road networks;

ii. A source-based model for predicting vehicle-derived contaminant loads (to include copper, zinc and PAH) in the identified `higher risk' road sections, based on emission factors from the MoT's research programme, and factoring in the effects of varying road/traffic conditions.

The vehicle contaminant load model developed under ii) above is described in Appendix E.

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Appendix E: Vehicle contaminant load model

E1 Introduction

Appendix E describes how the vehicle contaminant load model (VCLM) for road runoff was derived in terms of source data, assumptions and derivation of formulae relating annual pollutant load to traffic and road characteristics.

A full description of the developed model is given in Section E4. This includes the four key model equations that provide an estimate of road contaminant loads for the four contaminants of interest (particulate matter, zinc, copper and PAH).

The extensive literature review conduct by Kennedy et al. (2002) on behalf of the Ministry of Transport (MoT) identified and described the five main sources of contaminants to road runoff from vehicles. These are:

1) Brake wear (see Section E3.3 of this appendix)
2) Tyre wear (see Section E3.4)
3) Oil leakage (see Section E3.5)
4) Exhaust emissions (see Section E3.6)
5) Road surface wear (see Section E3.7)

Each of the above sources is described below in terms of the nature of contaminants, emission rate, composition and derivation of equations linking emissions rate to road/traffic characteristics.

A summary of the model equations for deriving contaminant loads in road runoff from vehicles prior to any stormwater containment or treatment device is described in Section E4.2. The means for how these loads may be attenuated by stormwater containment or treatment devices and the removal efficiencies that have been built into the model for the contaminants of interest are discussed in Section E4.4.

E2 Road deposition and retention factors

Sections E3.3 to E3.7 of this appendix describe how the model was developed to estimate the mass load of particulate and other contaminants released by the various vehicle-related sources.

Once generated, the pollutants are subject to a range of physical processes that are important since they determine the amount of contaminant deposited onto the road surface and therefore available for transport by road runoff. Moncrieff & Kennedy (2004) provide a detailed account of the processes and pathways that determine the fate of contaminants released from road vehicles.

These processes (e.g. wind dispersion) may reduce the amount depositing or remaining on the road surface prior to wash off by rainfall. This, in turn, reduces the quantity of material available for uptake in stormwater runoff. To determine the actual load available for road runoff the output must be corrected for losses using two factors.

The first factor is the proportion of contaminants discharged from the vehicle (or by the vehicle movement) that is deposited on the road surface. This is termed the `road deposition factor' and this is discussed below for each of the main contaminant sources. The factors used have been taken from Section 9 of Kennedy et al. (2002) which contains a full description of their magnitude and variability.

The second factor to consider is the proportion of contaminants that are retained on the road surface after being deposited from the vehicle. This is termed the road retention factor. For the purposes of this model, and in the absence of definitive data in the literature, a conservative road retention factor of 1 has been adopted for all pollutants.

It is noted that there is considerable uncertainly in the values to apply for both the above factors, as they will be seasonally dependent and site specific. However, this will not affect use of the model in terms of a screening tool for comparative purposes to identify priority areas in terms of contaminant load, as the same loss factors are assumed to apply to all areas.

E3 Derivation of model equations

E3.1 Brake wear

E3.1.1 Introduction

The two principle types of vehicle brakes, disc and drum, both have components that wear down during normal operation. The disc brake pads and the linings and shoes used in drum brakes (collectively known as friction linings), deteriorate with the action of braking, releasing particles onto the road surface and into the air. The emission rate for contaminants from brake wear is derived from the brake lining wear rate and the brake particle composition, as discussed below.

E3.1.2 Brake lining wear rate

Indicative particle emission rates from the wear of brake linings are provided by Kennedy et al. (2002) for a range of braking intensity and vehicle type (Table E3.1). The Average Wear Rate is derived from data given in Table 3.1 of Kennedy et al. (2002) which is a summary of average wear rates for motor vehicles based on an extensive international literature review of a range of vehicle types (excluding New Zealand as no local data were available). A wide variation is found for each vehicle type and hence the average rate in Table E3.1 is an order of magnitude indication only.

The indicative emission rates corresponding to Low, Moderate and Intense brake use were derived by Kennedy et al. (2002) by multiplying the Average Wear Rate by factors of 0.5, 1.5 and 2, respectively. As the authors note, these factors are arbitrary but are intended to provide a spread in break pad wear rates that reflects the known variations attributable to different driving conditions.

Table E3.1 Indicative particle emission rates from brake lining wear (mg/VKT)[16].

Braking Intensity	Passenger Car	Light Duty Vehicle	Heavy Duty Vehicle
Average Wear Rate	21	30	80
Low Brake Use	10.5	15	40
Moderate Brake Use	31.5	45	120
Intense Brake Use	42	60	160

Application of these emission rates is based on the braking situation, dictated by the road and traffic conditions (Table E3.2). Increasing brake use is a function of braking frequency and/or the intensity of brake use. The road terrain, dominant driving condition and level of service (e.g. interrupted or congested) will dictate if moderate or intense braking is applied. Low brake use is assumed as the default within the model. For a road with a mixture of conditions, the highest applicable braking rate is applied. For example a flat, central urban road that is congested will use the emission rate corresponding to intense brake use.

Classification of road terrain, road type and traffic condition is defined in Annex E2.

Table E3.2 Brake use as a function of road/traffic condition.

Brake use	Road/traffic condition
Intense	Terrain = Mountainous *or* Traffic Condition = Congested
Moderate	Terrain = Hilly *or* Road Type = Central Urban *or* Traffic Condition = Interrupted
Low	All other conditions

E3.1.3 Brake particle composition

Average estimates of friction lining composition from relatively limited New Zealand data were presented in Kennedy et al. (2002) as a means of determining the contaminants generated from brake wear.

As the authors note, the very large variation in both the type of friction linings and their composition makes it difficult to derive a meaningful average concentration for contaminants from brake lining wear. This is compounded by the fact that dust derived from brake wear may have different composition from the parent material. For example, Kennedy et al. (2002) cite a huge variation in copper in brake pads in New Zealand from a sample set of 39 that varies from 29 mg/kg (10 percentile) to 116,000 mg/kg (90 percentile). The authors recommend an interim value of 5000 ppm.

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[16] Kennedy et al. (2002) page 20 Table 3.7.

Table E3.3 details the indicative concentration of the key contaminants from New Zealand brake pad data that are used in the road runoff model to derive loading estimates. As noted above, the values are subject to large uncertainty and are considered to be order of magnitude.

Table E3.3 Indicative concentration of contaminants in friction lining particles.

Contaminant	Average concentration (mg/kg)
Zinc[17]	1630
Copper[18]	5000
Total PAH[18]	16

E3.1.4 Road deposition factor

International studies (e.g. Rogge et al. 1993) have shown that up to 11% of dust from brake lining wear will be trapped in drum brake housing, while a much smaller proportion is expected to be retained in vehicles with disc brakes.

Kennedy et al. (2002)[19] note that approximately 30% of particles emitted from brake lining wear become airborne and are subsequently lost from the road corridor. Based on this early study, they proposed an interim road deposition factor of 0.7 to account for the proportion of brake wear particles available for road runoff. This figure has also been adopted in the road runoff model.

E3.1.5 Derivation of equations

The particulate emission rates (Table E3.1), assumed deposition factor and the breakdown of vehicle kilometres travelled (VKT) by vehicle type (from the MoT's Vehicle Fleet Emission Model - see Annex E1) have been combined. This allows the following equations to be derived for particulate emission rates (mg/VKT) as a function of brake use and vehicle type for estimating contaminant loads in road runoff:

$$\text{Low Brake Use (mg/VKT)} = 0.7 \times [11 \times (1 - \%HCV) + 40 \times \%HCV \qquad \text{(Equation E3.1)}$$

$$\text{Moderate Brake Use (mg/VKT)} = 0.7 \times [33 \times (1 - \%HCV) + 120 \times \%HCV] \qquad \text{(Equation E3.2)}$$

$$\text{Intense Brake Use (mg/VKT)} = 0.7 \times [45 \times (1 - \%HCV) + 160 \times \%HCV] \qquad \text{(Equation E3.3)}$$

Where: %HCV = Percentage Heavy Commercial Vehicles (>=3.5tonnes) in traffic flow

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[17] Kennedy et al. (2002) page 21 Table 3.8, based largely on Kennedy & Gadd (2003).

[18] Kennedy et al. (2002) page 22 Table 3.9, based on Rogge et al. (1993).

[19] Kennedy et al. (2002) page 85, based on Cha et al. (1983).

These equations may be combined and simplified to give:

Brake Lining Particulate (mg/VKT) = 7.7 x Bi + 20. x %HCV x Bi *(Equation E3.4)*

Similarly, equations can be derived for the other contaminant indicators, utilising the indicative concentrations in Table E3.3:

Brake Lining Zinc (ng/VKT) = 12551 x Bi + 3308 x %HCV x Bi *(Equation E3.5)*

Brake Lining Copper (ng/VKT) = 38500 x Bi + 101500 x %HCV x Bi *(Equation E3.6)*

Brake Lining PAH (ng/VKT) = 123.2 x Bi + 324.8 x %HCV x Bi *(Equation E3.7)*

Where:
%HCV = Percentage Heavy Commercial Vehicles (>=3.5tonnes) in traffic flow
Bi = Brake intensity factor (see Table E3.4 below)

Table E3.4 Brake intensity factor.

Brake use[20]	Brake intensity factor, *Bi*
Low	1
Moderate	3
Intense	4

E3.2 Tyre wear

E3.2.1 Introduction

Frictional forces between a vehicle tyre and the road surface results in abrasive tyre wear. This generates tyre particulate matter which, if deposited on the roadway, contributes to the particulate and constituent contaminant loading in road runoff.

The key factors influencing tyre wear rate are the nature of the road surface and driving conditions. Tyre brand (design and composition) is a further significant factor.

The emission rate for contaminants from tyre wear is derived from the tyre wear rate and tyre particle composition, as discussed below.

E3.2.2 Tyre wear rate

Kennedy et al. (2002) proposed simplified tyre particulate emissions for varying vehicle types and driving conditions. These rates were generated from industry-supplied

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[20] Refer to Annex E2 for brake use classifications.

estimates of wear over the tyre lifetime, assumed tyre life, number of tyres on each vehicle and overly simplified factoring to account for different traffic conditions.

Unfortunately, despite detailing New Zealand estimates, the tyre wear rates in Kennedy et al. (2002) were based on a foreign estimate of private car tyre life. In addition, the wear rates per tyre for medium and heavy commercial vehicles were calculated using the estimated lifetime of tyres for private cars (a factor of 2.5 lower than heavy trucks). Thus, unlike the rates for other sources, the final tyre wear rates recommended within the Kennedy et al. (2002) report have not been incorporated in this model. The basic methodology for calculating average tyre wear from total wear and tyre lifespan has, however, been employed.

Average tyre wear rates (in mg/VKT) for different vehicle types were calculated by multiplying the total wear over the tyre lifetime (W in kg/tyre) by the number of tyres (N) and dividing by the estimated life of the tyres (L in km):

$$\text{Average Tyre Wear (mg/VKT)} = (W \times N/L) \times 10^6 \qquad \text{(Equation E3.8)}$$

The resulting average wear rates by vehicle type for this calculation, together with supporting data, are presented in Table E3.5.

Table E3.5 Average tyre wear rate by vehicle type.

Vehicle type	Size	Wear over lifetime, W (kg/tyre)[21]	Estimated tyre life, L (km)[22]	Assumed number of tyres, N	Average wear rate (mg/VKT)
Cars	-	1.1	47,000	4	94
LCVs	-	1.8	47,000	4	153
HCVs	Small	7.5(a)	119,000(a)	6	378
-	Medium	7.5(a)	119,000(a)	8	504
-	Large	7.5(a)	119,000(a)	12	756

(a) Reference quotes `heavy truck' or `heavy truck tyre' undifferentiated by size of HCV.

As set out by Kennedy et al. (2002), research has shown that tyre wear varies with a range of factors dependent on how the tyre interacts with the road surface. These include road surface, tyre composition, vehicle characteristics and route and style of driving. The most significant factor, especially with respect to this road runoff model, is the `route and style of driving', as these determine the frequency and magnitude of acceleration.

Different styles of driving can result in wear rates that vary by a factor of six. The route factor can result in an order of magnitude variation in tyre wear depending on whether the road section is straight or windy. Kennedy et al. (2002) quote variations in wear

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[21] Kennedy et al (2002) page 31, based on New Zealand industry estimates of material loss over tyre service life.

[22] Kennedy et al (2002) page 30, based on New Zealand data from Carpenter & Cenek (1999).

rates that vary from straight and level driving (representing 100% of tyre life) to 76% on slightly hilly and curvy roads, to 50% on hilly and curvy roads. The higher wear results from increased acceleration and breaking. Similarly, roads with interrupted flows (e.g. caused by an intersection where the driver brakes before accelerating away) will result in higher tyre wear than free flow conditions.

Since acceleration and braking are linked, the criteria used to classify tyre wear conditions as a function of road/traffic conditions are the same as for brake use (see Table E3.6). Annex E2 details how the road definitions are applied.

Table E3.6 Tyre wear condition as a function of road/traffic condition.

Tyre wear condition	Road/traffic condition
Intense Tyre Wear	Terrain = Mountainous *or* Traffic Condition = Congested
Moderate Tyre Wear	Terrain = Hilly *or* Road Type = Central Urban *or* Traffic Condition = Interrupted
Low Tyre Wear	All other conditions

E3.2.3 Tyre particle composition

As with brake (friction) linings, there is a large variation in New Zealand in both the types of tyre and their composition and hence the composition of particles derived from tyre wear. A summary of elemental composition of a selection of 12 tyres available in New Zealand[23] covering cars, LCVs and trucks indicated levels of zinc varying from 1,190 ppm – 18,300 ppm, and trace levels of copper in the range <1 to 3ppm. Data for PAH concentrations in tyres were obtained from international literature.

A summary of interim tyre compositions suitable for use in estimating contaminant emission loads was presented in Kennedy et al. (2002). Rates for the three key contaminants are reproduced in Table E3.7.

Table E3.7 Indicative concentrations of contaminants in tyre particles.

Contaminant	Concentration (mg/kg)
Zinc [24]	8310
Copper[25]	1
Total PAH[25]	226

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[23] Kennedy et al. (2002) page 34 Table 4.8.

[24] Kennedy et al. (2002) page 38 Table 4.13.

[25] Kennedy et al. (2002) page 38 Table 4.14, based on Rogge et al. (1993).

E3.2.4 Road deposition factor

Cadle & Williams (1978) showed that particulate from tyre wear was largely independent of wear rate and together with gaseous emissions accounted for up to 20% of the total mass lost. The balance was considered to be large particles that would settle close to the road. The study also indicated that less than 5% of wear product reaching the road surface was capable of being re-suspended.

Based on this work, Kennedy et al. (2002) suggested a preliminary factor of 0.8 to account for tyre wear products that are deposited to the road environment. This figure has also been adopted in the road runoff model.

E3.2.5 Derivation of equations

Combination of the average tyre wear rates (Table E3.5), assumed road deposition factor and the breakdown of VKT by vehicle types (from the MoT's Vehicle Fleet Emission Model - Annex 1), gives the following equations for tyre suspended solids present in road runoff:

$$\text{Average Tyre Wear (mg/VKT)} = 0.8 \times [103 \times (1 - \%HCV) + 556 \times \%HCV] \qquad \text{(Equation E3.9)}$$

Where: %HCV = Percentage Heavy Commercial Vehicles (>=3.5 tonnes) in traffic flow

The above equation can be rearranged and factored to take into account varying tyre wear conditions, producing:

$$\textbf{Tyre Particulate (mg/VKT)} = 82.4 \times Ti + 362.4 \times \%HCV \times Ti \qquad \text{(Equation E3.10)}$$

Similarly, equations can be derived for the other key contaminants utilising the concentrations in Table E3.7:

$$\textbf{Tyre Zinc (ng/VKT)} = 684744 \times Ti + 3011544 \times \%HCV \times Ti \qquad \text{(Equation E3.11)}$$

$$\textbf{Tyre Copper (ng/VKT)} = 82.4 \times Ti + 362.4 \times \%HCV \times Ti \qquad \text{(Equation E3.12)}$$

$$\textbf{Tyre PAH (ng/VKT)} = 18622.4 \times Ti + 81902.4 \times \%HCV \times Ti \qquad \text{(Equation E3.13)}$$

Where:
%HCV = Percentage Heavy Commercial Vehicles (>=3.5 tonnes) in traffic flow
Ti = Tyre wear intensity factor (see Table E3.8 below)

Table E3.8 Tyre wear intensity factor.

Tyre wear[26]	Tyre wear intensity factor, *Ti*
Low tyre wear	0.2
Moderate tyre wear	1.6
Intense tyre wear	3.3

E3.3 Oil leakage

E3.3.1 Introduction

Small leakage of oil occurs from vehicles through normal operation. These droplets of oil fall onto the roadway and are washed into the highway drainage system by road runoff. The emission rate for contaminants from oil leakage is derived from the oil loss rate and oil composition, as discussed below.

E3.3.2 Oil loss rate and oil composition

Kennedy et al. (2002) provide a general discussion on the nature of lubricant oil loss from vehicles and the typical compositions of used oils and this reference has been used to provide source data for these attributes. The oil loss rate cited in this reference is taken from the HDM-4 model described in Bennet & Greenwood (2001) and equates to 2.8 mL/1000 km for cars and most LCVs, and 2.1 mL/1000 km for HCVs.

The concentration of metals in used oil was sourced by Kennedy et al. (2002) from tests made by Zi ba-Palus (1998) on fresh and used Castrol GTX motor oil using data from a vehicle that had travelled 4000 km (about half way through the oil change cycle). This type of oil is commonly used in New Zealand vehicles.

The PAH concentration of used oil in cars in the above reference was taken from the middle of the range (560 mg/kg) found for a test on a petrol car by Kingett Mitchell (1994). The used oil was at the end of the oil change cycle and the PAH value is therefore likely to be an overestimate. The equivalent PAH concentration in used oils for HCVs (344 mg/kg) was taken by Kennedy et al. (2002) from tests on petrol and diesel cars (Kingett Mitchell 1994).

E3.3.3 Contaminant emission rates from oil leakage

Kennedy et al. (2002) combined the oil loss rates and oil composition data (referred to above) to generate contaminant emission rates for oil leakage to roads (see Table E3.9). This information is used as a basis for input to the road runoff model.

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[26] Refer to Annex E2 for tyre use classifications.

Table E3.9 Key contaminant emission rates from oil leakage (µg/VKT).

Contaminant	Cars and LCVs	HCVs
Zinc (Zn)[27]	2.9	2.1
Copper (Cu)[28]	0.0025	0.0019
Total PAH[28]	1.4	0.64

Oil leakage is assumed to have negligible impact on particulate contamination in stormwater runoff. Emission factors for this source are likely to be order of magnitude estimates only as no allowance is made for different driving conditions, speeds or age of vehicle.

E3.3.4 Road deposition factor

Due to the nature of oil leakage it is assumed that the full contaminant load reaches the stormwater runoff. In practice it is likely that a proportion remains adsorbed within the road surface and is slowly leached by stormwater or physically removed by tyre wear.

Kennedy et al. (2002)[29] suggested an interim factor of 1.0 for deposition of oil and associated contaminants that are leaked to the road environment. This figure has also been adopted in the road runoff model.

E3.3.5 Derivation of equations

Rates within Table E3.9 can be placed into the following equations:

$$\text{Oil Leakage Zinc (ng/VKT)} = [2.9 \times (1 - \%HCV) + 2.1 \times \%HCV] \times 1000 \quad \text{(Equation E3.14)}$$

$$\text{Oil Leakage Copper (ng/VKT)} = [0.0025 \times (1 - \%HCV) + 0.0019 \times \%HCV] \times 1000 \quad \text{(Equation E3.15)}$$

$$\text{Oil Leakage PAH (ng/VKT)} = [1.4 \times (1 - \%HCV) + 0.64 \times \%HCV] \times 1000 \quad \text{(Equation E3.16)}$$

Where: %HCV = Percentage Heavy Commercial Vehicles (>=3.5 tonnes) in traffic flow

These equations can be rearranged to give:

$$\textbf{Oil Leakage Zinc (ng/VKT)} = 2900 - 800 \times \%HCV \quad \text{(Equation E3.17)}$$

$$\textbf{Oil Leakage Copper (ng/VKT)} = 2.5 - 0.6 \times \%HCV \quad \text{(Equation E3.18)}$$

$$\textbf{Oil Leakage PAH (ng/VKT)} = 1400 - 760 \times \%HCV \quad \text{(Equation E3.19)}$$

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[27] Kennedy et al. (2002) page 46 Table 5.8, based on Zi ba-Palus (1998).

[28] Kennedy et al. (2002) page 47 Table 5.9, based on Kingett Mitchell (1994).

[29] Kennedy et al. (2002) page 86.

E3.4 Exhaust emissions

E3.4.1 Introduction

Combustion of fuels in a vehicle engine generates exhaust gases and particulate matter. If these emissions deposit on the roadway, or are washed out of the atmosphere by rain, they enter the road's stormwater runoff, and contribute to the contaminant load.

The methodology used to derive emission rate for contaminants from exhaust emissions is discussed below.

E3.4.2 Particle emission rates

There have been extensive studies conducted on the nature of particle emissions from vehicle exhausts. Most of these have focussed on fine particulate (PM_{10}) as this size fraction presents the greatest health risk from inhalation into the lungs. However, PM_{10} behaves like a gas and is mixed by turbulence and dispersed downwind from the road corridor. Emission rates for larger particulate matter from vehicle exhausts, which are likely to deposit on the road surface and contribute to road runoff, are generally not available. Consequently the discussion below relates to PM_{10} data.

The MoT Vehicle Fleet Emissions Model (VFEM) produces rates for particulate matter (PM_{10}), expressed for four different road types and three traffic conditions (see Table E3.10). These rates are weighted averages for the typical vehicle operating in the New Zealand fleet in 2001. Classification of road type and traffic condition is defined in Annex E2. (Note: due to the minor contribution of exhaust emissions to stormwater runoff, separate rates for heavy and light vehicles were not considered necessary.)

Table E3.10 PM_{10} emission rates (g/km) from the MoT VFEM[30].

Road type	Free flow	Interrupted	Congested
Central Urban	0.215	0.267	0.339
Motorway	0.223	0.158	0.173
Rural	0.172	0.166	0.198
Suburban	0.185	0.208	0.253

E3.4.3 Contaminant emission rates from exhaust emissions

Kennedy et al. (2002) concluded from a review of the literature that there was insufficient published data to establish clear relationships between the mass of PM_{10} emitted and the concentration of contaminants in particulate matter from vehicle exhausts. As such, emission rates for zinc, copper and PAH in exhaust emissions, as assumed by Kennedy et al. (2002), are independent of the rates of emission of PM_{10} (and consequently particulate in road runoff).

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[30] VFEM rates for 2001 vehicle fleet (model as at October 2005).

Table E3.11 lists the values cited by Kennedy et al. (2002) which have also been adopted in the road runoff model.

Table E3.11 Key contaminant emission rates from exhaust emissions (µg/VKT).

Contaminant	Light vehicles	Heavy vehicles
Zinc	46[31]	620[32]
Copper[32]	15	88
Total PAH[33]	916	2358

The rates for total PAH are derived from a subset of hydrocarbons listed in Table 7.22 of Kennedy et al. (2002). The listed rates for all contaminants are indicative as published emission data for such chemical species are extremely variable between studies.

E3.4.4 Road deposition factor

Due to the rapid dispersal of exhaust emissions, most contaminants released from this source do not deposit on the road surface directly from the roadway. Nevertheless, a fraction of particulate from vehicle emissions dispersed in the atmosphere may subsequently be washed out to the roadway by precipitation, along with other suspended matter.

Kennedy et al. (2002) identified an interim road deposition factor of 0.05 for particles associated with vehicle exhaust emissions[34] and this factor has been adopted in the road runoff model.

E3.4.5 Derivation of equations

Equations for contaminant loading in stormwater runoff can be derived from Tables E3.10 and E3.11:

$$\text{Exhaust Particulate (mg/VKT)} = 0.05 \times E \times M \times 1000 \qquad \text{(Equation E3.20)}$$

$$\text{Exhaust Zinc (ng/VKT)} = 0.05 \times [46 \times (1 - \%HCV) + 620v\%HCV] \times 1000 \qquad \text{(Equation E3.21)}$$

$$\text{Exhaust Copper (ng/VKT)} = 0.05 \times [15 \times (1 - \%HCV) + 88 \times \%HCV] \times 1000 \qquad \text{(Equation E3.22)}$$

$$\text{Exhaust PAH (ng/VKT)} = 0.05 \times [916 \times (1 - \%HCV) + 2358 \times \%HCV] \times 1000 \qquad \text{(Equation E3.23)}$$

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[31] Kennedy et al. (2002) page 75 Table 7.21, based on Gertler et al. (2002).

[32] Kennedy et al. (2002) page 75 Table 7.21, based on Norbeck et al. (1998).

[33] Kennedy et al. (2002) page 75 Table 7.22, based on Norbeck et al. (1998).

[34] Kennedy et al. (2002) page 87.

Where:

EPM = Exhaust PM_{10}, selected from Table E3.10

%HCV = Percentage Heavy Commercial Vehicles (>=3.5 tonnes) in traffic flow

Rearranging these equations gives:

$$\textbf{Exhaust Particulate (mg/VKT)} = 50 \times EPM \qquad \text{(Equation E3.24)}$$

$$\textbf{Exhaust Zinc (ng/VKT)} = 2300 + 28700 \times HCV \qquad \text{(Equation E3.25)}$$

$$\textbf{Exhaust Copper (ng/VKT)} = 750 + 3650 \times \%HCV \qquad \text{(Equation E3.26)}$$

$$\textbf{Exhaust PAH (ng/VKT)} = 45800 + 72100 \times \%HCV \qquad \text{(Equation E3.27)}$$

E3.5 Road surface wear

E3.5.1 Introduction

Tyres wear due to the frictional forces between them and the road surface. Conversely, the road surface is also abraded by the movement of vehicles, producing additional particulate and associated contaminant loading.

The emission rate for contaminants from road surface abrasion is derived from the wear rate and road surface composition, as discussed below. Since the vast majority of traffic to be assessed by this model will be travelling on bitumen/aggregate sealed roads, analysis has been limited to this road type.

E3.5.2 Road surface wear rate

Kennedy et al. (2002) proposed a methodology for calculating road surface (bitumen and aggregate) wear based on design life[35] :

$$BW = \frac{W \times D \times \%B \times \rho_B \times 10^5}{Yd \times 365 \times AADT} \qquad AW = \frac{W \times D \times \%A \times \rho_A \times 10^5}{Yd \times 365 \times AADT}$$

(Equations E3.28 and E3.29)

Where:

BW = total bitumen wear (g/VKT)

AW = total aggregate wear (g/VKT)

W = width of wear, cm (assumed to be 100cm)

D = depth worn, cm (assumed to be 0.85cm)

%B = % of wear composed of bitumen (assumed to be 50%)

%A = % of wear composed of aggregate (assumed to be 50%)

ρ_B = density of bitumen (assumed to be 1 g/cm^3)

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[35] Kennedy et al. (2002) page 82.

ρ_A = density of aggregate (assumed to be 2.7 g/cm^3)
Yd = design life in years
AADT = annual average daily traffic on the road section

With the above assumed values, the wear of bitumen and aggregate simplify to:

$$BW = \frac{11644}{Yd \times 365 \times AADT} \quad \text{(Equation E3.30)}$$

$$AW = \frac{31348}{Yd \times 365 \times AADT} \quad \text{(Equation E3.31)}$$

The calculation of road seal design life, Yd, is based on the number of layers present in the seal [36]:

For single coat seals:

$$Yd = 4.916 + 1.68ALD - (1.03 + 0.219ALD)\log(elv) \quad \text{(Equation E3.32)}$$

For multilayer seals:

$$Yd = 14.87 + ALD - 3.719\log(elv) \quad \text{(Equation E3.33)}$$

Where:

Yd = design life in years
elv = equivalent light vehicles/lane/day
ALD = average least dimension of the sealing chip in mm used on the section (for multilayer seals the larger ALD is used)

$$elv = \frac{AADT}{Lanes}\left(1 + \frac{9(\%HCV)}{100}\right) \quad \text{(Equation E3.34)}$$

Where:

AADT = annual average daily traffic on the road section
Lanes = number of lanes across whole road
%HCV = percentage heavy commercial vehicles

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[36] Transit New Zealand Specification TNZ P/17:2002.

E3.5.3 Road surface composition

Bitumen wear particles contain a range of organic and inorganic substances that may be liberated into road runoff. Table E3 12 details the concentration in bitumen of the key contaminants considered in this model. (Note: worn aggregate is considered inert and therefore does not contribute to the concentration of contaminants in stormwater.)

Table E3.12 Concentration of contaminants in bitumen wear particles[37].

Contaminant	Concentration (mg/kg)
Zinc	54
Copper	46
Total PAH	10

E3.5.4 Road deposition factor

Wind (local and vehicle derived turbulence) will transport a portion of road wear particles outside the road corridor. Nevertheless, Kennedy et al. (2002) made the conservative assumption that all road wear material is available for stormwater runoff[38] and this deposition factor of 1.0 has also been adopted in the road runoff model.

E3.5.5 Derivation of equations

Bitumen wear is likely to create fine particles capable of becoming suspended solids within the stormwater. The character of aggregate wear particles is more diverse, with a mixture of a fine fraction, due to the erosion of the aggregate itself, and entire aggregate pieces, that are released with the removal of bitumen binder. Thus, a proportion of aggregate ‘particles’ will be incapable of suspension in the runoff. However, the intact aggregate fragments can create a loading on the stormwater system, especially treatment devices such as catchpits.

With limited research available regarding the exact nature of road surface wear, it was assumed that the total wear rate of both bitumen and aggregate contributed to the loading of solid material in road runoff. This conservative approach will tend to overestimate the suspended solid load in storm runoff.

However, due to the nature of the design life equation, this methodology will produce a highly exaggerated estimate of road wear pollution loading for carriageways with less than 2,000 AADT. It is proposed that road wear be ignored for streets with less than 2,000 vehicle movements per day.

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[37] Kennedy et al. (2002) page 83, based on Kennedy & Gadd (2003).

[38] Kennedy et al. (2002) page 87.

On this basis, the combined aggregate and bitumen wear rate equations above are summed to give (for AADT >= 2,000):

$$\textbf{Road Surface Particulate Matter (mg/VKT)} = \frac{43082000}{Yd \times AADT} \quad \text{(Equation E3.35)}$$

Combination of the bitumen wear rate and the contaminant concentrations in Table E3.12 produces the following equations (for AADT >= 2,000):

$$\textbf{Road Surface Zinc (ng/VKT)} = \frac{628776000}{Yd \times AADT} \quad \text{(Equation E3.36)}$$

$$\textbf{Road Surface Copper (ng/VKT)} = \frac{535624000}{Yd \times AADT} \quad \text{(Equation E3.37)}$$

$$\textbf{Road Surface PAH (ng/VKT)} = \frac{116440000}{Yd \times AADT} \quad \text{(Equation E3.38)}$$

E4 Model description

This section provides a summary description of the vehicle contaminant load model for road runoff (`the model') under the following subsections:

Section E4.1 - Model overview
Section E4.2 - Contaminant load model equations
Section E4.3 - Contaminant load model inputs
Section E4.4 - Treatment of road runoff
Section E4.5 - User inputs
Section E4.6 - Worked examples

Benefit values (willingness to pay) for removal of the four key contaminants from stormwater, thus providing a common unit of comparison for ranking purposes, are included in Section E4.4.

An initial validation of the model using published data is given in Section E5.

E4.1 Model overview

The model is an Excel-based programme which estimates total mass loadings (kg/VKT) of specific vehicle-derived contaminants in road runoff. The pollutants are particulate, zinc, copper and polyaromatic hydrocarbons (PAH) as these are the primary vehicle-derived contaminants of interest in terms of road runoff. Figure E4.1 gives a schematic of the model process.

The contaminant load per VKT is calculated using the overall contaminant load model equations given in Section E4.2. By multiplying the contaminant loading per VKT by the road section AADT and length, the model creates an estimate of the total annual load of contaminants in runoff generated on that section of road prior to any containment or treatment device.

Depending on the treatment device or stormwater containment method in place, the contaminant loading reaching any waterbody can be significantly reduced. Contaminant load in the outfall of the device is a function of the quantity of flow treated and the contaminant reduction efficiency. Multiplying the total road contaminant load by these removal rates (detailed in Section E4.4) gives an approximation of the annual load of contaminants in road runoff i.e. discharged from the road network outfall.

(Note: this load may undergo further dilution/attenuation along a discharge *pathway* before it impacts a sensitive receiving environment; alternatively a direct impact may occur when the load is discharged directly to a sensitive waterbody. The *pathway attenuation factor* is to be considered in a subsequent stage of this research project.)

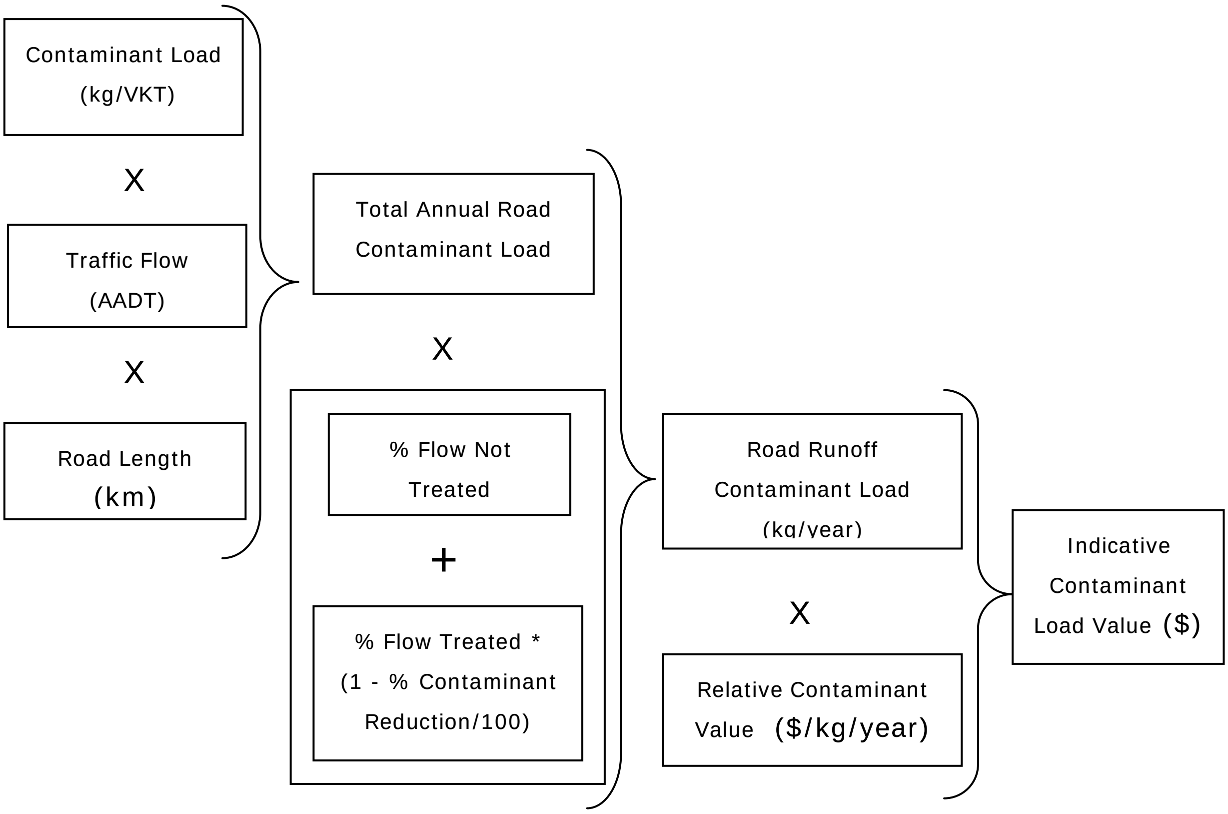

Figure E4.1 Schematic of contaminant load calculation.

Contaminants produce a range of environmental impacts and, as described in Section E4.4, the perceived value that the community places on these impacts varies. By multiplying the individual road runoff contaminant loads in runoff by their perceived financial values, and summing the contributions, an indicative contaminant loading value is derived. This concept has potential value for ranking road or road networks in terms of pollutant load but its application and use have not been further developed under this project.

The model is designated a `vehicle contaminant load model' as it is restricted to emissions from road vehicles – it does not quantify pollutants in road runoff that arise from other sources e.g. open land, urban or industrial discharges and which contribute to the overall stormwater pollutant load. However, the model is capable of distinguishing the relative contributions of contaminant loads from different road types (e.g. state highways *vs.* local roads within a catchment) that may affect a given receiving environment.

E4.2 Contaminant load model equations

The equations derived for each vehicle-based contaminant source (detailed in Section E3) have been combined to produce the four key model equations used for estimating contaminant loads in road runoff for particulate matter (PM), zinc, copper and PAH:

Where AADT < 2,000:

$$PM(mg/VKT) = 7.7Bi + 82.4Ti + \%HCV[20.3Bi + 362.4Ti] + 50 \times EPM \quad \text{(Equation E4.1)}$$

$$Zinc(\mu g/VKT) = 5.2 + 12.5Bi + 684.7Ti + \%HCV[33.1Bi + 3011.5Ti + 29.9] \quad \text{(Equation E4.2)}$$

$$Copper(\mu g/VKT) = 0.753 + 38.5Bi + 0.0824Ti + \%HCV[101.5Bi + 0.362Ti + 3.649] \quad \text{(Equation E4.3)}$$

$$PAH(\mu g/VKT) = 47.2 + 0.123Bi + 18.6Ti + \%HCV[0.325Bi + 81.9Ti + 71.3] \quad \text{(Equation E4.4)}$$

Where AADT >= 2,000:

$$PM(mg/VKT) = 7.7Bi + 82.4Ti + \%HCV[20.3Bi + 362.4Ti] + 50 \times EPM + \frac{4308200}{Yd \times AADT} \quad \text{(Equation E4.5)}$$

$$Zinc(\mu g/VKT) = 5.2 + 12.5Bi + 684.7Ti + \%HCV[33.1Bi + 3011.5Ti + 29.9] + \frac{628776}{Yd \times AADT} \quad \text{(Equation E4.6)}$$

$$Copper(\mu g/VKT) = 0.753 + 38.5Bi + 0.0824Ti + \%HCV[101.5Bi + 0.362Ti + 3.649] + \frac{535624}{Yd \times AADT} \quad \text{(Equation E4.7)}$$

$$PAH(\mu g/VKT) = 47.2 + 0.123Bi + 18.6Ti + \%HCV[0.325Bi + 81.9Ti + 71.3] + \frac{116440}{Yd \times AADT}$$

(Equation E4.8)

Where:

%HCV = Percentage Heavy Commercial Vehicles (>=3.5 tonnes) in traffic flow
Bi = Brake intensity factor
Ti = Tyre wear intensity factor
EPM = Exhaust Particulate Matter (PM_{10})
Yd = Design life in years
AADT = Annual average daily traffic on the road section

E4.3 Contaminant load model inputs

The inputs required for the contaminant load model, together with supporting equations, are summarised below:

Table E4.1 *Bi* : Brake Intensity Factor.

Brake use[39]	Brake intensity factor, *Bi*
Low	1
Moderate	3
Intense	4

Table E4.2 *Ti* : Tyre Wear Intensity Factor.

Tyre wear[40]	Tyre wear intensity factor, *Ti*
Low tyre wear	0.2
Moderate tyre wear	1.6
Intense tyre wear	3.3

Table E.3 EPM: PM_{10} Emission Rates (g/km) from the MoT VFEM[41].

Road type	Free flow	Interrupted	Congested
Central Urban	0.215	0.267	0.339
Motorway	0.223	0.158	0.173
Rural	0.172	0.166	0.198
Suburban	0.185	0.208	0.253

For single coat seals: $Yd = 4.916 + 1.68ALD - (1.03 + 0.219ALD)\log(elv)$

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[39] Refer to Annex 2 for brake use classifications.

[40] Refer to Annex 2 for Tyre use classifications.

[41] VFEM rates for 2001 vehicle fleet, model as at October 2005.

For multilayer seals: $Yd = 14.87 + ALD - 3.719\log(elv)$

Where:

Yd = design life in years
elv = equivalent light vehicles/lane/day
ALD = average least dimension of the sealing chip in mm used on the section (Note: for multilayer seals the largest value of ALD is used.)

$$elv = \frac{AADT}{Lanes}\left(1 + \frac{9(\%HCV)}{100}\right)$$

Where:

AADT = annual average daily traffic on the road section
Lanes = number of lanes
%HCV = percentage heavy commercial vehicles

E4.4 Treatment of road runoff

E4.4.1 Introduction

The equations described in Section E4.2 enable an estimate to be made of the contaminant load (i.e. particulate matter, zinc, copper and PAH in mg/year) deposited on a road surface from a given set of traffic and road attributes.

However, the actual contaminant load that leaves the highway in road runoff will be determined by the nature of the highway drainage infrastructure, including any treatment devices. The type of drainage infrastructure (e.g. swale or kerb/channel) plays a key role in controlling the pollutant load and therefore the risk from contaminants in stormwater runoff.

This section describes how the runoff model takes account of the existing (or proposed) stormwater treatment device in order to estimate the contaminant load that is actually discharged to the receiving environment.

E4.4.2 Stormwater treatment devices

Hartwell & Welsh (2005) conducted a detailed literature review of the most common stormwater containment and treatment devices that are used for road runoff. A list with a description of each of these treatment devices is given in Annex E3. Further details on the design guidelines for each type are to be found in ARC (2003).

E4.4.3 Treatment efficiencies

Hartwell & Welsh (2005) also included a review of the contaminant retention (removal) efficiency of a range of stormwater treatment devices. A summary is given in Table E4.4.

The flow treatment figures are approximate and refer to the percentage of influent by that is treated by the device. It was assumed that the reduction rates for PAH are the same as those reported by Hartwell & Welsh (2005) for TPH. This is a reasonable assumption as PAH are a subgroup of Total Petroleum Hydrocarbons (TPH).

The removal efficiencies for the four contaminants provide the typical mass reduction that is achievable for the device, assuming that this is well maintained and correctly sized for the expected throughput. These rates have been used in the runoff model to take account of any treatment device on the section of highway in estimating pollutant loads discharged to the receiving environment from the road corridor.

The flow treatment figures are approximate and refer to the percentage of influent by that is treated by the device. It was assumed that the reduction rates for PAH are the same as those reported by Hartwell & Welsh (2005) for Total Petroleum Hydrocarbons . This is a reasonable assumption as PAH are a subgroup of TPH.

Table E4.4 Approximate retention efficiency of various stormwater treatment devices[42].

Device No.	Treatment device	Annual flow treated %	Retention efficiency total mass (%)			
			Suspended solids	Zinc	Copper	PAH [43]
1	Culvert/direct discharge	100	0	0	0	0
2	Catchpits (average) [44]	100	64	29	36	10
3	Dry Water Quality Pond	100	63	27	41	10
4	Wet Water Quality Pond	100	77	36	51	10
5	Wet Extended Detention Pond	100	80	41	54	10
6	Constructed Wetland	100	77	54	69	10
7	Swale	100	75	47	57	47
8	Bio-retention Device (80% rain garden; 20% swale)	100	83	59	62	65
9	Rain Garden	90	84	51	63	48
10	Proprietary Device Type 1(a) - Gross Pollutant Traps	74	30	9	18	10
11	Proprietary Device Type 2(b) - Filtration Systems	90	84	44	59	48
12	Proprietary Device Type 3(c) - Catchpit Filter Systems	90	42	13	25	10

Note: (a) Examples include Downstream Defender, CDS, Ecosol, CleansAll; (b) Examples include Sand Filter, Storm Filter; (c) Examples include Enviropod, Ecosol 100, FloGuard

The removal efficiencies for the four contaminants provide the typical mass reduction that is achievable for the device, assuming that this is well maintained and correctly sized for the expected throughput. These rates have been used in the runoff model to take account of any treatment device on the section of highway in estimating pollutant loads discharged to the receiving environment from the road corridor.

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[42] From Hartwell & Welsh (2005) p36 Table 5.2, unless stated otherwise.

[43] Listed values are for Total Petroleum Hydrocarbons and are assumed the same for PAH.

[44] Derived from Timperley et al. (2005) and Ng et al. (2003).

E4.4.4 Relative contaminant values

Hartwell & Welsh (2005) produced benefit values based on the reduction of individual contaminants in road runoff by various stormwater treatment devices. This was derived from a Contingent Valuation approach to determine willingness-to-pay and impact weightings based on survey questionnaires of communities in Rodney District, Waitakere City and North Shore City. The survey found that 44% of households were in favour of treating runoff from roads at a cost of $1,000 per household per year. The benefit values for unit reduction of each of the key contaminants (in $/kg/year) are given in Table E4.5.

Table E4.5 Weighted benefit values for contaminant reduction[45].

Contaminant	Benefit value $/kg/year
Suspended solids	44
Zinc	37,801
Copper	144,818
PAH[46]	106,871

Of the four contaminants considered, the lowest benefit is derived for suspended solids and the highest is for copper. Reduction of copper has a perceived benefit of almost four times that of zinc. Caution is needed in application of these benefit values. As the authors point out, TPH was highly weighed by the public, possibly “because oils are very visual and easy to connect these to roads. It is harder for the public to make the link between roads and metals such as zinc and copper”.

With these limitations in mind, the benefit values can be used to add together the differing emission loads of the four key contaminants derived by the runoff model to provide an overall benefit for the road network. The runoff model includes the benefit value calculation for ranking and comparative purposes only.

(Note: the monetary benefit derived from these values does not equate to the cost of stormwater treatment for the relevant road section; it is merely the community’s perceived value expressed in monetary terms of stormwater treatment in improving the receiving environment.)

E4.5 User inputs

User-specified inputs that are required to run the model are listed in Table E4.6. The column `RAMM Table’ indicates the equivalent designation of each variable (where appropriate) in the RAMM database. The column `Modification’ refers to instructions on how to adjust the RAMM input data to run the model or suit the user preference.

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[45] Hartwell & Welsh (2005) p39 Table 6.3.

[46] Values for Total Petroleum Hydrocarbons, and assumed the same for PAH.

Table E4.6 User inputs to the model.

Variable name	RAMM table[(a)]	Modification
AADT (vehicle/day)	Traffic [use ADT]	None
Road Type	Carriageway	Allocate RAMM classifications to road type groups defined in Annex E2
Level of Service (LoS)	-	Calculate and apply groups defined in Annex E2
HCV (%)	Carriageway	None
No of Lanes	Carriageway	None
Road Length (km)	Carriageway	None
Horizontal Terrain (deg/km)	High speed geometry	Convert radians to degrees via trigonometry
Vertical Terrain (m/km)	High speed geometry	Convert units
No of Seal Layers	Surface structure	Allocate RAMM information as either single or multilayer
ALD (mm)	Surface structure	None
Treatment	Drainage	Insert device number (Table E4.1) or assign nearest device to road section - alternative treatment devices may need to be stipulated from local knowledge

(a) New Zealand's Road Asset Maintenance Management system

E4.6 Worked examples

Table E4.4 contains three worked examples of the road runoff contaminant load model. These illustrates both the model input and output data and range of contaminant loads for the four main pollutants – particulate matter, zinc, copper and PAH.

E4.6.1 Example A

Example A compares the effects of varying traffic flows on a typical suburban road under three increasing Levels of Service: 10,000 AADT (free flow), 15,000 (interrupted) and 20,000 (congested). All other variables remain constant. The runoff is discharged through a catchpit.

The example demonstrates the increase in contaminant load as a function of traffic flow. However the relationship is not linear. A doubling of traffic from 10,000 to 20,000 AADT increases the copper discharge by a factor of about 7 and the zinc discharge by a factor of about 30. This is due to the increased level of congestion with the higher traffic flows and consequent increase in vehicle braking, acceleration and tyre wear.

E4.6.2 Example B

This example compares the effects of pollutant load on two different road types (suburban road and a motorway), both carrying the same level of traffic. The suburban road has 2 lanes while the motorway has six. All other variables remain constant. The runoff is discharged without treatment.

In this case the heavy metal contaminant load in the road runoff is very similar. The slight reduction for motorway values is due to a lower equivalent light vehicles per lane per day (`elv' term in Equation E3.34) and hence a slightly higher design life for the road (`Yd' term in Equation E4.6 for zinc, and Equation E4.7 for copper). This results in a small reduction in surface wear of the motorway compared with the suburban road. The particulate matter load for the two road types is similar but in this case the difference in PM_{10} emission rates for motorway and suburban roads (`EPM' term in Equation E4.5) has a further contributory effect. In practice these differences are negligible.

E4.6.3 Example C

Example C compares the effects of varying road conditions (bends and hills) on the contaminant load for two sections of rural highway, one hilly and the other mountainous (see terrain classification in Table A2.3), but each with identical levels of traffic and %HCVs. All other variables remain constant. The runoff is discharged without treatment.

The example demonstrates the increase in contaminant load as a function of more mountainous terrain with a larger number of tighter bends causing more braking (as seen by 32% more copper) and much greater tyre wear (as seen by a doubling of the zinc). Other pollutants also increase in the rugged mountain section compared with the hilly road.

E4.6.4 Example D

The final example compares the contaminant loads from identical roads with and without treatment. The suburban road with 10,000 AADT and a catchpit (column 1) is compared with the same road with no stormwater treatment device (column 4), all other factors being identical.

The example demonstrates the importance of catchpits in removing suspended sediment and heavy metals. The average catchpit has an approximate retention efficiency of 64% for suspended solids (see Table E4.7 with corresponding figures for copper of 36% and zinc 29%). Retention of PAH is low at only 10%.

Table E4.7 Vehicle contaminant load model - input and output data for example road scenarios.

Attribute	Example A (Change in traffic flow)			Example B (Change in road type)		Example C (Change in road condition)	
Model Input	1	2	3	4	5	6	7
Road Type	Suburban	Suburban	Suburban	Suburban	Motorway	Rural Highway	Rural Highway
Level of Service (LoS)	1: Free Flow	2: Interrupted	3: Congested	1: Free Flow	1: Free Flow	1: Free Flow	1: Free Flow
AADT (veh/day)	10,000	15,000	20,000	10,000	10,000	10,000	10,000
HCV (%)	10%	10%	10%	10%	10%	15%	15%
No of Lanes	2	2	2	2	6	2	2
Road Length (km)	1	1	1	1	1	1	1
Horizontal Terrain (deg/km)	10	10	10	10	10	55	250
Vertical Terrain (m/km)	25	25	25	25	25	55	65
No of Seal Layers	1	1	1	1	1	1	1
ALD (mm)	8.5	8.5	8.5	8.5	8.5	8.5	8.5
Treatment	Catchpit	Catchpit	Catchpit	None(a)	None(a)	Culvert	Culvert
Model Output							
Particulates (g/day)	1,981	3,182	5,226	5,502	4,811	7,675	10,107
Zinc (mg/day)	1,623	17,447	47,270	2,286	2,275	18,878	38,372
Copper (mg/day)	359	1,457	2,555	561	552	1,690	2,230
PAH (mg/day)	551	1,332	2,594	612	610	1,092	1,619
Relative contaminant value[47]	1	10.7	29.0	1.4	1.4	11.6	23.6

(a) No treatment device - direct discharge to receiving environment (equivalent to 'culvert' in Table E4.4).

Example D (see text) compares columns 1 and 4.

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[47] Based on Weighted benefit value of contaminant reduction, Table E4.5.

E5 Model validation

E5.1 Introduction

This section provides an overview of the initial validation of the vehicle contaminant load model described above using published data. A review is initially given of emission rates derived in New Zealand for vehicle-derived pollutants in road runoff. The model is then used to predict emission factors for copper and zinc using input data from the Richardson Road (Auckland) study.

E5.2 Review of NZ data on vehicle emission factors

A number of studies have been conducted in New Zealand to estimate emission factors for vehicle-derived pollutants that affect road runoff.

Snelder (1995) estimated the loading rates of vehicle-derived contaminants to impervious surfaces in New Zealand urban environments and from this estimated the loading rates to urban waterbodies.

Kennedy & Gadd (2003) described a road dust sampling study in Waitakere City designed to provide field validation data for the VFEM-W model. The emission factors in the VFEM-W model were used to predict loads for the Waitakere City roads. The measured contaminant loads were also used by these authors to back-calculate emission factors based on the road length and traffic volume.

More recently, NIWA conducted a detailed monitoring programme in order to develop a model for metal loads contributed to urban stormwater from road runoff. The site chosen was a 0.5km section on Richardson Road, west Auckland (Timperley et al. 2003). The concentrations of metals (dissolved and particulate) were measured and related to traffic flow in order to calculate an emissions factor per vehicle-kilometre (VKT).

The emission factors (mg/VKT) for copper, zinc and PAH from the above studies are summarised in Table E5.1. The emission factors are variable across the studies. Emission factors for copper are generally within a factor of two. Emission factors for zinc are more variable with the VFEW-W emission rate about 7 times higher than the rate back-calculated from the Waitakere field study. Likewise, the VFEM-W factor for PAH is about 5 times higher than the other quoted values.

The variability between measured and predicted emission rates in the Waitakere City study is mainly attributed to large uncertainties within the source-based VFEM-W model. This is particularly the case for the wear rate of brakes and the average composition of brake pads (for copper), and the wear rate of tyres and their average composition (for zinc), for vehicles in New Zealand (Kennedy and Gadd, 2003).

Table E5.1 Contaminant emission factors for road runoff from various New Zealand studies.

Study	Emission factors (mg/vkt)		
	Copper	Zinc	PAH
Snelder 1995, NZ urban waterways	0.16	0.7	0.015
Kennedy & Gadd 2003, Waitakere City [48]	0.16	0.18	0.014
Kennedy & Gadd 2003, VFEM-W [49]	0.086	1.2	0.079
Timperley et al. 2003, Richardson Rd	0.0593	0.447	-

E5.3 Data used for model validation

As discussed above, several studies have been conducted in New Zealand to assess the contribution of roads to stormwater quality. Unfortunately, most of these studies are based on the empirical modelling method, discussed in Appendix D3.1. As such, differentiation and recording of key attributes of importance to source based assessments are not included in these reports. Therefore, the data available for validation of this model is sparse.

The viable data available for validation is limited to the detailed monitoring of Richardson Road in Auckland (Timperley et al. 2003 and 2005). This study considered a 0.5 km section of road is a single carriageway carrying approximately 17,000 vehicles per day with a well-defined surface drainage system discharging to a single point that was monitored.

The attributes and assumed values (inputs) from the Richardson Road study that were used to test the contaminant load model described in this paper are given in Table E5.2.

The horizontal and vertical terrain components were estimated from a photograph of the Richardson Road site (Figure 6, Timperley et al. 2005). For purposes of this calculation, and in the absence of actual data, the HCV fraction was assumed to be a nominal 10% (national average figure). Although the true value is likely to be not dissimilar to this assumed value, the pollutant model is sensitive to %HCV and therefore this will introduce some uncertainty in the derived pollutant loads.

A major uncertainty in the Richardson Road study was the average *in situ* retention efficiency of catchpits for metals in suspended sediment, as the efficiency drops as sediment builds up in these devices. In the initial study, Timperley et al. (2003) pre-cleaned the catchpits, which maximised the retention efficiencies. The measured efficiencies are therefore higher than would be expected compared with the Auckland-wide `average' catchpit partially filled with sediment. (For total sediment, the authors

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[48] Median value back-calculated from measured loads (individual data show large variability).

[49] Normal driving conditions based on average NZ vehicle fleet (values increase for more congested conditions).

Table E5.2 Modelled attributes and values from Richardson Road study.

Attribute	Assumed value
AADT	17,354
Road Type	SU
LoS	2: Interrupted
% HCV	10%
# of Lanes	2
Horizontal Terrain (degrees/km)	<50
Vertical Terrain (m/km)	<45
# of Seal Layers	1
ALD (mm)	8.5
Treatment	Catchpit (average)

estimated that this would equate to about 75% retention compared with an *in situ* figure of about 60%.)

In the subsequent study (Timperley et al. 2005), reduced retention efficiencies were developed for the more realistic situation of partially filled catchpit (the `average' catchpit). The pollutant loads actually entering the stormwater system for these `average' catchpits are correspondingly higher.

The catchpit retention efficiencies used in this model validation are those given in Table E4.1 for an `average' catchpit. These equate to a partially filled catchpit derived from data in Ng et al. (2003) and Timperley et al. (2005).

E5.4 Model predictions compared with field study

Contaminant load rates (in mg/VKT for total copper and total zinc) derived from applying the contaminant load model to the Richardson Road site data are given in Table E5.3. These are compared with results for total metals (sum of dissolved and particulate fractions) reported in Timperley et al. (2005), assuming an average catchpit efficiency from their model predictions at the same site. Also shown are data for the VFEM-W model (Kennedy & Gadd 2003).

A direct comparison may be made between predictions for this model and the Timperley model, both assuming an `average' catchpit (i.e. one in use and partially filled with sediment), as shown by data highlighted (*) in Table E5.3. The emission factors for copper are within a factor of two while the emission factors for zinc are more divergent (within a factor of four). In both cases, the model in this study overestimates the contaminant load compared with Timperley et al. (2005).

As discussed in Section D3.1, the contaminant load model developed in this project (like the MoT's VFEM-W model) is a source-based model developed from vehicle emission

Table E5.3 Comparison of model predictions of emission factors for Richardson Road, Auckland.

Study	Total copper (mg/VKT)	Total zinc (mg/VKT)
This Model - average catchpit LoS = 1	0.05	0.23
This Model - average catchpit LoS = 2	0.15 *	1.6 *
This Model - average catchpit LoS = 3	0.20	3.3
Timperley et al. 2005 [50] - average catchpit	0.078 *	0.45 *
Timperley et al. 2005 [51] - clean catchpit	0.055	0.368
VFEM-W (normal driving conditions - LoS = 1) [52]	0.086	1.2
VFEM-W (congested traffic – LoS = 3)[52]	0.12	2.3

Note: LoS = Level of Service (i.e. degree of traffic congestion – see Table A2.2).

* data comparable

factors derived from the MoT's research programme, and based on source components (e.g. zinc from tyres based on wear rates and tyre composition) and traffic/road characteristics. The Timperley model is empirical where contaminant loads are derived from correlating the measured concentrations of metals in road runoff to road traffic data under actual field conditions. The results in Table E5.3 compare favourably, given the different approaches used and the uncertainties in assumptions of both models.

The emission factors derived from the model (this study) are particularly sensitive to the Level of Service (degree of traffic congestion). For example, copper shows a fourfold increase in emission rates in moving from normal flows (LoS 1) to congested conditions (LoS 3). The increase for zinc is even more marked (a factor of 14 for the same comparison). The Level of Service has been estimated at a value of 2 for purposes of comparing model outputs but the actual value may be less than 2. This limitation could be overcome by replacing the discrete LoS bands with a continuous function relating emission factors to vehicle speed.

The findings indicate that further refinement and calibration with field data is required before the model may be used to estimate absolute pollutant loads in runoff from roads. Notwithstanding these uncertainties, preliminary validation of the vehicle pollutant load model developed in this study shows that it will find application in later stages of the research project for comparative assessment of contaminant loads from different road networks.

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[50] Timperley et al. (2005), p31, estimated for partially filled catchpits (reflecting likely *in situ* sediment retention efficiency).

[51] Timperley et al. (2005), p29 Table 4, pre-cleaned catchpits (as per study).

[52] Kennedy & Gadd (2003) Table 4.2.

E6 Conclusions and recommendations

E6.1 Conclusions

The following conclusions are drawn on the vehicle contaminant load model developed in this study:

- A contaminant load model for road runoff has been developed for vehicle-derived pollutants (copper, zinc, PAH and suspended solids). The model is based largely on emission factors developed and published under the MoT's research programme on effects of transport on the aquatic environment. The model allows user selection of stormwater treatment devices as part of the existing highway drainage.

- Initial validation of the model for copper and zinc has been achieved by applying the model to published field data and road characteristics for the Richardson Road site in Auckland. The correlation between the predicted pollutant load rates from the model (mg/VKT) is within a factor of two (copper) and four (zinc). In both cases the model overestimates the contaminant load.

- The model output is particularly sensitive to the Level of Service (traffic congestion) as this is currently incorporated in the model as three discrete bands. A continuous function relating emission factors to vehicle speed would overcome this limitation.

- There is considerable uncertainly in the derivation of emission factors in the model, given the wide variability in source data, hence the model predictions must be treated as only order of magnitude estimates of contaminant load.

- Based on these inherent limitations and uncertainties, the model should be used to estimate contaminant loads from road networks on a comparative rather than absolute basis.

E6.2 Recommendations

It is recommended that the following considerations be taken into account in development of the risk-based SRE screening methodology that comprises the overall objective of this project:

1) A simple screening of road networks based on VKT by sub-catchment be considered in combination with criteria being developed for SRE sensitivity under Stage 2 to identify sections of road with higher risk potential (Tier 1 screening process).

2) The pollutant load model described in this report is applied to the identified higher risk road sections to provide a more definitive and comparative measure of the risk

from road traffic, for taking forward in the SRE risk assessment process (Tier 2 assessment).

3) Further validation and sensitivity analysis of the contaminant load model is undertaken to assess the limits of its application for estimating absolute contaminant loads (outside the scope of this project).

E7 References

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Annex E1 VKT split by vehicle type

An estimate of the VKT proportional splits for different vehicle types was taken from the MoT's Vehicle Fleet Emissions Model (VFEM), Table EA1.

The VFEM's development has been dormant since 2002, and VKT estimates beyond 2001 are highly speculative. Hence, values for 2001 were used in this model. If the VFEM is updated any improved data should be incorporated in this model.

Table EA1 VKT splits by vehicle type (2001).

Vehicle	Size	% of VKT	% of VKT subtotals
Motorcycles	-	0.35	-
Cars	-	76.93	-
LCVs	-	13.00	-
Total Light	-	-	90.28
HCVs	Small	4.03	-
-	Medium	1.66	-
-	Large	4.03	-
Total Heavy	-	-	9.72

Annex E2 Definitions for road type, traffic condition and terrain

EA2.1 Definitions

The following section gives definitions for:

- Road Type
- Traffic Condition (Level of Service)
- Terrain

EA2.1.1 Road Type

The road type categories utilised in the model are outlined in Table EA2.1. Roads should be assigned to the category that best exemplifies its nature. Accuracy in this area is not critical to the operation of the model: the road type input is used primarily for the selection of PM_{10} emissions, a minor contributor to suspended solid loading. The default of 'SU' should be used in urban areas if further classification is not possible.

Table EA2.1 Road type.

Abbreviation	Name	Description[53]
MO	Motorways	Essentially urban or urban fringe; characterised by limited access design, grade separated intersections, multi-lane in a relatively wide road reserve (includes urban State Highways).
CU	Central Urban Roads	Central business areas, high frequency of signalised intersection control and relatively low speeds, retail and office land use in close proximity.
SU	Suburban Roads	Urban/suburban arterial/collector/distributor/local access roads other than CU and MO.
RH	Rural Highways and local Roads	Rural highways: State highways, predominantly 2 lane main connecting routes; relatively high percentage of heavy traffic, some 3 lane sections on gradients and where horizontal geometry is restricted. Rural local roads: local authority, 2 lane, mainly distributor and local access in nature.

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[53] Ministry of Transport (1998).

EA2.1.2 Traffic Condition: Level of Service

Level of Service (LoS) is defined as the ratio of traffic volume to roadway capacity and is a measure of the degree of road traffic congestion. AADT or ADT is used as a measure of traffic volume. The capacity of the road can be devised using the criteria in Transfund's Project Evaluation Manual [commencing in section A3.11[54]]. Definitions of the Level of Service categories used in the vehicle contaminant load model are given in Table A2.2.

Table EA2.2 Definition of Level of Service (LoS).

Category	Name	Definition
1	Free Flow	$\frac{AADT}{Capacity} \leq 0.35$
2	Interrupted	$0.35 < \frac{AADT}{Capacity} < 0.7$
3	Congested	$\frac{AADT}{Capacity} \geq 0.7$

EA2.1.3 Terrain

Terrain is classified as per Transfund's Project Evaluation Manual and as summarised in Table EA2.3.

Table EA2.3 Terrain classification[55].

Vertical terrain (m/km)	Horizontal terrain (degrees/km)	Classification
< 45	< 50	Flat
> 45	< 50	Rolling
< 45	50 < x < 150	Rolling
> 45	50 < x < 150	Hilly
< 60	150 < x < 300	Hilly
> 60	150 < x < 300	Mountainous
All	> 300	Mountainous

The absolute values of a road's vertical terrain, contained in the RAMM system, should be averaged and a unit conversion applied prior to application in this classification process. Horizontal terrain is recorded in RAMM as the radius of the road's curvature and must be converted to degrees before use.

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[54] Transfund (2004).

[55] Derived from Transfund (2004) Table A10.5.

Table EA2.4 Brake use as a function of road/traffic condition.

Brake use	Road/traffic condition
Intense	Terrain = Mountainous *or* Traffic Condition = Congested
Moderate	Terrain = Hilly *or* Road Type = Central Urban *or* Traffic Condition = Interrupted
Low	All other conditions

Table EA2.5 Tyre wear as a function of road/traffic condition.

Tyre wear condition	Road/traffic Condition
Intense tyre wear	Terrain = Mountainous *or* Traffic Condition = Congested
Moderate tyre wear	Terrain = Hilly *or* Road Type = Central Urban *or* Traffic Condition = Interrupted
Low tyre wear	All other conditions

Annex E3 Stormwater transport and treatment devices

Table EA3.1 Stormwater transport and treatment devices[56].

Device	Description
Culvert (transport device)	A short closed (covered) conduit that passes stormwater runoff under an embankment, usually a roadway.
Note: The descriptor `culvert' is used in the runoff model for any drainage device (e.g. kerb and channel, open conduit, surface water channel or similar) that is used to transport water and which has no effect on the flow or water quality. The remaining items in the table are treatment devices.	
Catchpit	Small chamber incorporating a sediment trap that accepts runoff before entering a reticulated stormwater system.
Water Quality Pond (Dry)	`Dry' ponds (also known as detention ponds) temporarily store runoff following heavy rainfall and discharge this later under controlled conditions. The primary mechanism for contaminant removal is sedimentation.
Water Quality Pond (Wet)	`Wet' ponds (also known as retention ponds) maintain a permanent pool of water and may allow for the controlled release of runoff. The primary mechanism for contaminant removal is sedimentation.
Constructed Wetland	Constructed wetlands consist of shallow vegetated pond areas that are only practicable where space is available for construction. Wetlands remove contaminants through a combination of mechanisms including sedimentation, aerobic digestion and adhesion of contaminants to vegetation.
Swale	Swales are wide shallow grassed channels normally located adjacent to roads but often separated by a section of verge. They use a combination of slow, shallow water flow and vegetation to remove contaminants from stormwater. They can be used in place of drainage pipes and to convey flood flows. Swales are most effective on gently sloping sites (1% - 5%). In general a width of 3 – 7 m is required to accommodate design requirements.
Rain Garden	Rain gardens are a form of filtration device that use plants and layers of media (e.g. mulch, planting soils, gravel) for contaminant removal. Treatment may also occur through infiltration of stormwater to the base of the rain garden, depending on the underlying soils. The filtration media is placed in layers within a small trench or hollow. Topsoil is placed on the surface and planted. Rain gardens can be incorporated into a landscaping plan. Catchment area served tends to be small (<1,000m^3).

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[56] Taken in part from Hartwell and Welsh (2005) p34-36, Table 5.1.

Table EA3.1 (continued) Stormwater transport and treatment devices.

Device	Description
Bioretention Device	80% Rain Garden & 20% Swale
Gross Pollutant Traps	These include floating booms, gratings and mesh inserts within catchpits and culverts. Several proprietary products are available that use a combination of hydraulic motion and sedimentation to remove contaminants. For example, Continuous Deflection Separation (CDS) devices work by using hydraulic motion to separate out and remove contaminants. Stormwater entering a CDS unit is kept in continuous motion as it flows around and through a series of screens. Floating objects are retained and collected on the surface while heavier pollutants settle into a chamber at the base of the unit.
Filtration Systems	These remove contaminants using filtration media. Sand filters are useful where space restrictions apply and they can be designed to take traffic loads. They usually comprise a concrete tank containing sand through which stormwater is filtered and often include a settling chamber for removal of coarse material followed by a tank containing the filter media. Finer materials are trapped or adhere to the filter media. They can generally only service a small catchment area. Various proprietary products such as the StormFilter ® are also available. This is a filtration system that uses cartridges filled with an array of media, selected to treat the specific pollutant loadings at each site.
Catchpit Filter Systems	Devices in this category include floating booms, gratings and mesh inserts (e.g. 'Enviropods') installed within culverts and catchpits.

Appendix F: Glossary

In this report the following terms have the indicated meanings:

Assimilative Capacity: The ability of a receiving environment to assimilate or 'absorb' contaminants without evidence of significant adverse effects.

Contaminant: Includes any substance (including gases, odorous compounds, liquids, solids and micro-organisms) or energy (excluding noise) or heat, that either by itself or in combination with the same, similar, or other substances, energy, or heat –

(a) When discharged into water, changes or is likely to change the physical, chemical, or biological condition of water; or
(b) When discharged onto land or into air, changes or is likely to change the physical, chemical, or biological condition of the land or air onto or into which it is discharged (Resource Management Act 1991).

Contaminant load: The mass of chemical passing a given point or being deposited in a given area in a given time e.g. an annual load could be expressed in kg/annum.

Depositional Receiving Environment: Receiving environment which is low energy (subject to minimal water movement) and which therefore tends to accumulate sediments and their associated contaminant load.

Dispersive Receiving Environment: Receiving environment which is high energy (subject to strong water flow, tidal movement or wave action) and which therefore tends to disperse and dilute the contaminants present in runoff.

Environment includes:-
(a) Ecosystems and their constituent parts, including people and communities; and
(b) All natural and physical resources; and
(c) Amenity values; and
(d) The social, economic, aesthetic, and cultural conditions which affect the matters stated in paragraphs (a) to (c) of this definition or which are affected by those matters.

Receiving Environment: An aquatic water body into which urban stormwater or road runoff is discharged. The *immediate* receiving environment is the area in the vicinity of the discharge outfall whereas *distant (or final)* receiving environments are further afield; contaminants may accumulate in either type but are more likely to accumulate to ecologically significant levels in immediate receiving environments. Receiving environments include rivers, lakes, wetlands, estuaries, harbours and the open coastline.

Sensitivity of Receiving Environment: The susceptibility or vulnerability of receiving environments to adverse effects on ecology, human uses or human values (including cultural values).