ROad runoff Pollution management and mitigation of environmental Risks

D2.1 A review of current knowledge on the vulnerability of European surface water and groundwater to road related pollution, together with a critique of related assessment tools

CEDR PROPER PROJECT

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Executive summary

This report is the first deliverable of WP2 (Assessing the vulnerability of European surface and ground water bodies to road runoff during the building and operating of roads) of the CEDR PROPER project. It commences with a review of the European and international literature on the vulnerability of surface waters and groundwaters to road related pollution, together with an assessment of highway pollutant prediction models to identify/predict the vulnerability of receiving waters to such discharges. This report compliments the first deliverable of WP1 (which reviewed data and models to predict the pollutant loads and concentrations generated by highway activities), and will inform the development of deliverables in WP3 (Sustainable assessment of measures and treatment systems for road runoff) and WP4 (Sustainable assessment of measures and treatment systems for road runoff during construction work).

Data reviewed in this report are from studies undertaken at sites which, between them, cover a wide variety of climatic and geographic circumstances, sampling and analytical protocols and experimental designs and test species. The sections on surface water vulnerability synthesises results from studies on fish, aquatic invertebrates, plants, fungi and bacteria and amphibians either as a group, or where data permits, at a species level. Sections on groundwater vulnerability collate and critique data on the contributions of soil versus sediments as pollutant transport pathways and their resulting impacts on groundwater quality. Particular attention is given to the impact of de-icing salts on groundwater bodies, and current limitations in understanding fundamental groundwater pollutant transportation processes are described. Models reviewed with regard to their potential to predict receiving water vulnerability are HAWRAT (UK), SELDM (USA), IMPACT (USA), MT-GA (USA/Israel), PREQUALE (Portugal) and Impact of AADT (USA). Each model is described and a matrix developed which supports their comparative assessment against a range of criteria including input variables, pollutants and limitations.

Whilst acknowledging the issue of incomplete data sets and challenges of integrating data from disparate studies, the need for National Road Administrations and environmental protection agencies to make decisions now - on when, where and how road runoff should be treated – is also recognised. As a contribution to meeting this need, the conclusions provide a brief overview of the evidence base associated with each of the following key questions:

- Does highway runoff impact on the ecological and/or chemical status of receiving waters?
- What sort of impacts have been reported?
- Is there a relationship between AADT and ecological impact?
- What are the key contaminants in highway runoff?

Based on the findings presented, the multiple interpretations of the term 'vulnerable receiving water' are identified and a CEDR PROPER definition of the term for use within future outputs is proposed. The report concludes with a series of recommendations for further work to inform future CEDR research agendas.

Acknowledgements

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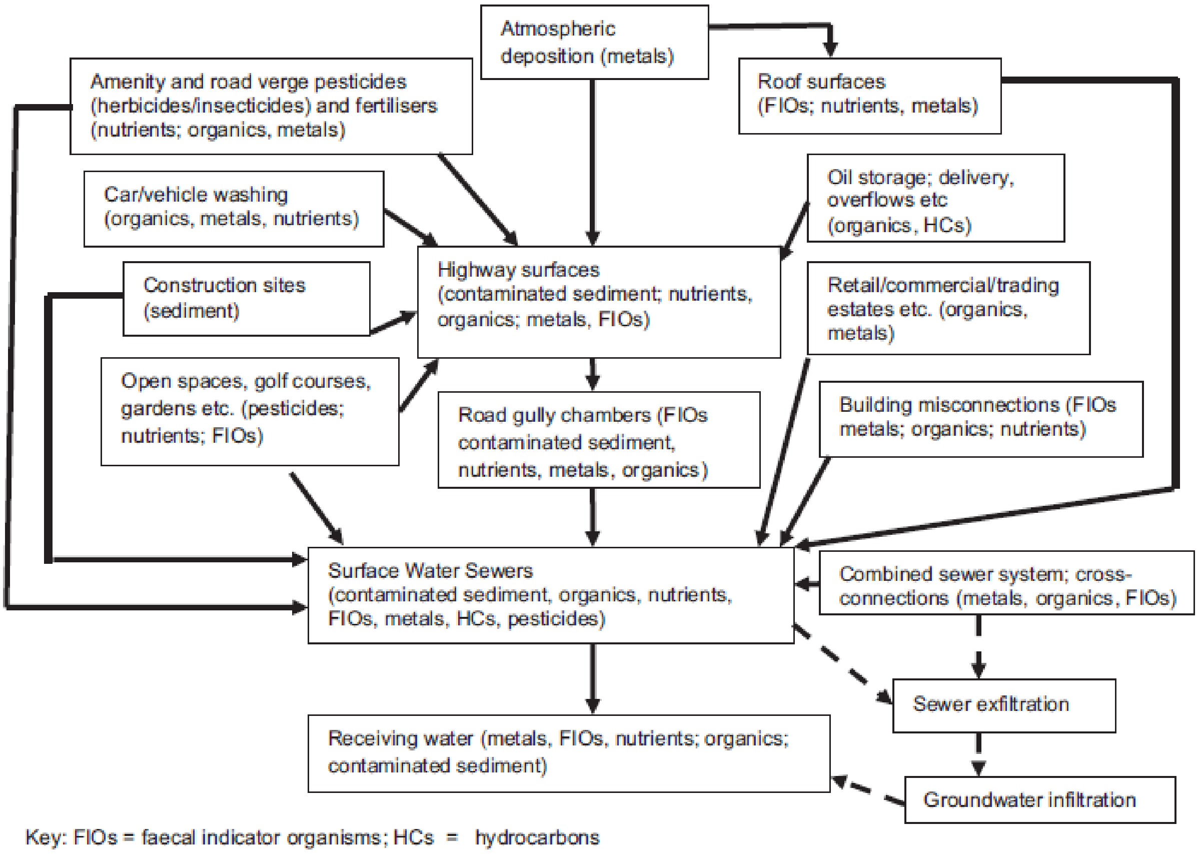
1. Introduction

The EU Water Framework Directive (EU WFD, 2000) establishes a legal obligation on Member States to protect and restore the quality of European water bodies, both surface waters and groundwaters. The main objective of the EU WFD is to achieve 'good status' for all of Europe's surface waters and groundwater by 2015 or 2027 at the latest and above all, to prevent deterioration of the existing status of a water body. 'Good status' for surface waters is defined through both ecological and chemical conditions in terms of a healthy ecosystem and low levels of chemical pollution. Groundwater status within the EU WFD is defined by whether there is sufficient water to maintain the health of the ecosystem it feeds to and assesses total abstraction against groundwater recharge. Groundwater chemical status is assessed separately through the evaluation of Annex II substances for the specific waterbody (EU Groundwater Directive, 2006). In the UK the point of compliance is defined as the unsaturated zone at a short distance in the direction of groundwater flow i.e. within the pollution plume. The methodological approach applied (in the UK at least) is a risk assessment approach based on a source-pathway-receptor (SPR) linkage where each component of the SPR linkage is identified and weighted to assess risks posed from surface recharge. The ecological status of surface waters is assessed in terms of the abundance of aquatic flora and fauna supported by the availability of nutrients and with regard to aspects such as salinity, temperature, water flow and volume and chemical pollution. The chemical status of surface waters is determined through the use of environmental quality standards (EQSs) which have been established for a range of chemical pollutants of concern (termed 'priority substances' and priority hazardous substances (EU Priority Substances Directive, 2013). This list consists of 45 regulated pollutants, which are considered to be bioavailable, toxic and persistent in the environment.

In order to fulfil the requirements of the EU WFD, Member States need to establish water quality objectives for water bodies and, where problems are identified, to propose appropriate mitigating measures. River Basin Management Plans (RBMPs) and accompanying Programmes of Measures (PoMs) explain these proposals and how they will be achieved within a given timeline. The overall objective is to protect the whole water body and to initiate a coordinated response to solve identified problems. Continued effort is required as it has recently been reported that 47% of the EU's surface waters had not achieved 'good ecological' status (EEA, 2014). Although there is some uncertainty associated with this figure, it has also been reported that impacts are categorised as being 'unknown' for 15% of the ecological status and 40% of the chemical status of surface water bodies (European Commission, 2012a). The unavailability of the necessary data and its associated implications has recently been highlighted in the case of Greece (Yannopoulos et al., 2013). To provide the flexibility to address these problems the WFD operates on a six year repeating cycle with the current cycle commencing in 2015. There will be a further review of progress in 2019 to consider the impact of the second round of RBMPs.

Whilst agricultural practices are recognised as a key pressure on water quality in many RBMPs, the role of roads and associated traffic as sources of diffuse pollution is less certain. For example, in the Thames river basin district (UK) it was reported that 17% of water bodies are affected by urban diffuse pollution (Thames RBMP, 2015). As a category, urban diffuse pollution includes runoff from a range of sources including roads, pavements, roofs and misconnections (see Figure 1.1), and therefore the exact contribution from traffic activities is not readily ascertainable. As a contribution to addressing this knowledge gap, this report compliments D1.1 (a review of the literature on the pollutant loads generated by road runoff under the varying climatic, environmental and road characteristics experienced throughout Europe) by undertaking a critical review of the current knowledge on the vulnerability of European surface waters and groundwaters to road-related pollution. This involved the assessment of 108 publications (scientific papers, national reports, case studies and research dissertations) undertaken within a range of European (e.g. UK, France, Germany, Norway, Sweden, Slovenia, Ireland, Portugal and Switzerland) and international (e.g. Japan, Canada and the USA) contexts. In addition, this report also evaluates a selection of desk-based tools used in the prediction of receiving water vulnerability in relation to a range of criteria including pollutants considered, input variables, methodology and limitations.



Key: FIOs = faecal indicator organisms; HCs = hydrocarbons

Figure 1.1 Principal sources and types of urban diffuse pollutants (Lundy et al., 2011)

Whilst it is fully recognised that, in addition to the EU WFD and EU Groundwater Directive, a further range of European and national legislation and international agreements are also relevant to mitigating the impact of road runoff on receiving waters, such measures are comprehensively addressed in Task 2.3 of this project and the reader is directed there for further information on pertinent legislation and policies. This report concludes with a synthesis of findings structured by key topics to facilitate use by National Road Administrations (NRAs), of the evidence-based identification of knowledge gaps and the recommendations of areas for further research. As such, this report makes a key contribution to supporting NRAs in playing a lead role in establishing the current situation in relation to the impact of road runoff on receiving surface and groundwaters (focus of WP2) and in identifying mitigation measures (focus of WP3).

2. Methodology

2.1 Literature review and assessment

Studies pertaining to the impact of road runoff on receiving waters were identified through undertaking searches of the science databases www.sciencedirect.co.uk and www.webofscience.co.uk using the key terms: road runoff, highway runoff, impact, vulnerability and risk assessment. The timeframe for papers was post-2000 – current but key papers preceding this date were also considered if they included key data sets / major findings. As well as contributing research papers from a national perspective, CEDR PROPER partners also sourced and submitted national studies, research dissertations etc. and members of the PROPER IAB were also asked to submit relevant studies from the grey literature. In total 108 references were received. Studies identified were reviewed and those in English related to the impact of road runoff on receiving waters are comprehensively discussed in Section 3 of this report.

Untreated highway runoff may be discharged directly to receiving rivers/streams or may be initially directed to a treatment system. These treatment systems may themselves present a surface water environment (e.g. retention basins, constructed wetlands, swales) but their chemical purpose is to reduce/remove highway pollutants and therefore they are not considered as natural water systems within this report. It is certainly true that these treatment systems can develop an active ecological status but because the focus of the PROPER project is to review the impacts of highway runoff on receiving rivers/streams, treatment systems have not been included in this literature review except in a few cases where a particularly important impact is addressed. However, these systems will be reviewed in WP3 of the CEDR PROPER project.

2.2 Impact database matrix

The impact database matrix (see CEDR PROPER website at: www.proper-cedr.eu/) consists of 60 discrete entries providing further catchment and supporting data on many of the studies assessed within Sections 3.1 and 3.2 of this deliverable. Data was entered into the matrix by various partners in varying levels of detail and all entries refer to papers and/or case studies which have explored or assessed the impact of road runoff on receiving waters. Whilst by no means an exhaustive review of the literature, in being compiled by active researchers and practitioners from a variety of EU Member States, the matrix does provide an overview of research undertaken within a range of climatic and environmental conditions. As well as a resource it in its own right, the matrix inputs were then synthesised by undertaking a simplified contents analysis which identified:

- the number of papers referring to an identified aspect e.g. infiltration rate
- the way data on the aspect was presented; for example, quantitatively (e.g. m/d; L/s) and/or qualitatively (e.g. moderate rate of infiltration)
- keywords to enable a user to locate specific papers (using the ctrl F function in Excel)

2.3 Vulnerability prediction model review and assessment

There are a considerable number and variety of modelling approaches that have been developed to predict impermeable surface pollutant concentrations and loadings and their potential impact on receiving waterbodies (e.g. MUSIC [EWater, 2012]; WinSLAMM [PV and Associated, 2017] and SWMM [EPA, 2006]). They vary from simplified approaches based on event mean concentration (EMC) values and storm event data (such as rainfall runoff depth) to multiple regression analysis, stochastic mass balances and complex artificial neural network analysis. Groundwater vulnerability modelling, such as that incorporated within DRASTIC and MODFLOW, is essentially based on flow velocities, travel times and dilution capacities in respect of individual pollutants. Many of the vulnerability models refer to short-term acute effects rather than long-term chronic impacts and few approaches consider detailed impacts of persistent or ultralow concentrations, or in-stream deposition and resuspension potential on ecological vulnerability.

Within this review, models specifically developed to address the concentrations and loads generated by highways are considered, and their ability to identify/predict the vulnerability of receiving waters to highway discharges evaluated. This involved the review of guidance manuals and studies reported in the literature pertaining to the following six different models:

- HAWRAT
- SELDM
- IMPACT
- MT-GA
- PREQUALE
- Impact of AADT

Of the models reviewed, only HAWRAT and SELDM specifically address an assessment of the risks posed to the receiving water environment. However, the remaining models are also briefly described as, they incorporate, to varying degrees, considerations of a range of processes/factors that influence the pollutant pathway from its source (i.e. the highway) to an identified receptor (e.g. surface water) and are therefore of value in terms of informing future work around predicting the magnitude of risk posed to receiving waters from highway discharges.

3. Chemical and ecological status of surface water and groundwater bodies near non-urban roads

3.1 Chemical/ecological impacts of highway runoff on surface waters

Due to the close relationships which exist between the pollutant levels in a receiving water and its ecological status, it is relevant that these aspects are considered together. This literature review will examine situations where the chemical status, expressed by environmental quality standards (EQS), is consistent with deteriorating ecological conditions. In contrast, there are instances where exceedance of EQS levels by certain pollutants is not accompanied by poorer ecological status. The difficulty clearly arises when a mixture of pollutants is present and it becomes difficult to establish a causal relationship between the chemical and ecological statuses. There is also the relative vulnerability of different biological species to consider and therefore this review is structured in sections, each addressing the ecological communities commonly found in receiving rivers/streams.

3.1.1 General impacts/WFD considerations

The EU Water Framework Directive (EU WFD, 2000) requires that 'good ecological and chemical status' should be achieved for all surface and groundwater bodies by initially 2015 but by 2027 at the latest. 'Good status' for surface waters is defined through both ecological and chemical conditions in terms of a healthy ecosystem and low levels of chemical pollution. The ecological status of surface waters is assessed in terms of the abundance of aquatic flora and fauna supported by the availability of nutrients, its hydro-geomorphology and with regard to aspects such as salinity, temperature, water flow and volume, and chemical pollution. The chemical status of surface waters is determined through the use of environmental quality standards (EQSs) which have been established for a range of chemical pollutants of concern (termed 'priority substances' and priority hazardous substances (EU Priority Substances Directive, 2013). The original list contains 33 regulated pollutants, which are considered to be bioavailable, toxic, and persistent in the environment (EU Environmental Quality Standards Directive, 2008). Table 3.1 identifies those pollutants from the revised and updated list of 45, which have been reported to be present in highway runoff, together with their current limit values according to acute/chronic toxic effects and the nature of the surface water. The EU has also published a ‘watch list’ of 8 substances for which an EU-wide monitoring exercise has been instigated to gather data to inform possible future legislation through the establishment of EQS values. In addition to the pollutants identified in the EU Priority Substances Directive (2013), there are specific standards which have been adopted by individual countries to protect ecological status e.g. copper and zinc within the UK.

The successful achievement of the 2027 deadline for the achievement of both good chemical and ecological status for surface waters will be dependent on the ability to control all contaminated inputs including highway runoff which represents a diffuse source of suspended solids, nutrients, salts, metals, and persistent organic pollutants. European member states will need to address these water quality issues when planning, building, and operating road networks (Meland 2016). At the planning and building stage, it is standard practice to carry out Environmental Impact Assessments (EIAs) or similar assessments to protect waters against pollution e.g. through the installation of appropriate treatment systems.

Once roads become operational, a guiding principle regarding the need to treat the runoff is often based on the annual average daily traffic (AADT) density with the critical limit being 10,000–15,000 vehicles/day (Meland, 2016). However, this is best used as a 'precautionary principle' as there is no sound evidence for the existence of a clear or consistent linear relationship between the number of vehicles and pollution loadings and/or concentrations in highway runoff. An alternative approach, developed in the UK, is the evidence-based Highways Agency Water Risk Assessment Tool (HAWRAT) which combines the ecological impacts of highway runoff with existing hydraulic conditions and the traffic characteristics (see Section 4.1). In order to assist in the achievement of the water protection requirements outlined in the WFD, a new common methodology has been proposed which defines the water body sensitivity to road pollution based on the intrinsic characteristics of inland (surface waters and groundwaters), transitional waters and coastal waters (Brenčič et al., 2012). The application of this methodology to two case-studies in Portugal and Slovenia allowed the classification of the case-study areas as either sensitive, non-sensitive or requiring further studies.

Table 3.1 Environmental Quality Standards for priority substances and priority hazardous substances likely to be identified in highway runoff.

CAS number	Name of substance	AA-EQS Inland surface waters (µg/L)	AA-EQS Other surface waters (µg/L)	MAC-EQS Inland surface waters (µg/L)	MAC-EQS Other surface waters (µg/L)
120-12-7	Anthracene*	0.1	0.1	0.4	0.4
71-43-2	Benzene	10	8	50	50
7440-43-9	Cadmium and its compounds* (depending on water hardness classes)	≤ 0.08 (Class 1)	0.2	≤ 0.45 (Class 1)	≤ 0.45 (Class 1)
		0.08 (Class 2)		0.45 (Class 2)	0.45 (Class 2)
		0.09 (Class 3)		0.6 (Class 3)	0.6 (Class 3)
		0.15 (Class 4)		0.9 (Class 4)	0.9 (Class 4)
		0.25 (Class 5)		1.5 (Class 5)	1.5 (Class 5)
330-54-1	Diuron	0.2	0.2	1.8	1.8
206-44-0	Fluoranthene*	0.1	0.1	1	1
7439-92-1	Lead and its compounds*	7.2	7.2	NA	NA
7439-97-6	Mercury and its compounds*	0.05	0.05	0.07	0.07
91-20-3	Naphthalene*	2.4	1.2	NA	NA
7440-02-0	Nickel and its compounds	20	20	NA	NA
NA	Polyaromatic hydrocarbons (PAH) *	NA	NA	NA	NA
50-32-8	Benzo(a)pyrene*	0.05	0.05	0.1	0.1
205-99-2	Benzo(b)fluor-anthene*	Σ = 0.03	Σ = 0.03	NA	NA
207-08-9	Benzo(k)fluor-anthene*				
191-24-2	Benzo(g,h,i)-perylene*	Σ = 0.002	Σ = 0.002	NA	NA
193-39-5	Indeno(1,2,3-cd)-pyrene*				
122-34-9	Simazine	1	1	4	4

NA = not applicable; * Priority hazardous substance
CAS = Chemical Abstracts Service
AA-EQS Environmental Quality Standard expressed as an annual average value; inland surface waters encompass rivers and lakes and related artificial or heavily modified water bodies.
MAC-EQS Environmental Quality Standard expressed as a maximum allowable concentration; when marked as 'not applicable', the values are considered protective against short-term pollution peaks in continuous discharges since they are significantly lower than the values derived on the basis of acute toxicity.

In addition to impacting on the receiving water environment, roads influence the local ecosystem through changes in soil density, temperature, soil water content, light levels and patterns of runoff. Not all species and ecosystems are equally affected by roads, but overall their presence is highly correlated with changes in species composition, population sizes, and hydrologic and geomorphic processes that shape aquatic and riparian systems (Trombulak and Frissell, 1999). The pattern of aquatic habitat loss differs from the terrestrial pattern yet nevertheless results in the ecological fragmentation of aquatic ecosystems. The movement of fish populations can be constrained by the existence of highway crossings, especially culverts, and highway networks can alter flow regimes within watersheds (Wheeler et al., 2005).

Wheeler et al. (2005) have compared the presence of a highway with the accompanying landscape urbanization and found that while they are of similar natures, characterized by physical and chemical impacts that are temporally persistent, the latter are of a greater magnitude and more widespread. The landscape urbanization stage is the greatest threat to stream habitat and biota, with stream ecosystems being sensitive to low levels (< 10%) of watershed urban development.

3.1.2 Biological monitoring

In order to establish a link between highway runoff discharges to receiving rivers/streams and any subsequent ecological effect, it is important to consider population and community techniques in conjunction with analytical chemistry determinations and other biological measures of contaminant stress. Water temperature is known to influence the metabolic and reproductive rates of algae, benthic invertebrates, and fish. Aquatic organisms are also sensitive to changes in dissolved oxygen, pH and alkalinity, and other water-quality properties and constituents. Therefore, a knowledge of watershed geochemistry is relevant to facilitate the interpretation of population and community data.

Different aspects of biomonitoring achieve separate but complimentary goals when population and community techniques are applied to different key species, including algae, benthic invertebrates, and fish. These different taxonomic groups respond differently to natural or anthropogenic disturbances because of differences in habitat, food, mobility, physiology, and life history. Thus, an approach that utilizes different species can provide information that can be used to develop cause-and-effect relationships.

3.1.2.1 Investigations involving a range of different species

Hurle et al. (2006) have carried out a comprehensive investigation of the effects of soluble pollutants contained in highway runoff on the ecology of receiving watercourses. As soluble pollutant loads are typically short in duration, the expected acute impacts on ecosystems have been assessed using 24-hour toxicity tests by exposing five fish species (brown trout [*Salmo trutta*], roach [*Rutilus rutilus*], bullhead [*Cottus gobio*], minnow [*Phoxinus phoxinus*] and 3-spined stickleback, [*Gasterosteus* aculeatus], 6 macroinvertebrates (*Baetis rhodani, Chironomus riparius, Erpobdella octoculata, Gammarus pulex, Hydropsyche pellucidula* and *Lymnaea peregra*) and 2 algae (*Sphaerium corneum, Selenastrum capricornutum*) to typical highway runoff and receiving water pollutant concentrations. The individual pollutants investigated include metals (copper, cadmium, zinc and aluminium), polycyclic aromatic hydrocarbons (PAHs, pyrene and fluoranthene), oils (diesel and crank case oil), de-icing agents and products of their breakdown (sodium chloride, potassium acetate, ammonia and cyanide) and herbicides (diuron and glyphosate). The objective was to derive, where possible, median (50 %) and 20 % lethal, effect or inhibition concentrations (e.g. LC50/LC20, EC50/EC20 or IC50/IC20). In a stream environment receiving highway runoff the pollutants may react or bind with e.g. particulates and dissolved organic matter, which can alter their bioavailability and/or toxicity and therefore laboratory toxicity tests can only give an indication of where potential ecological problems may exist.

The laboratory tests have identified that two metals, copper and zinc (but not cadmium), have the potential to be toxic to many of the tested species with fish being most at risk in soft water streams. However, fish were shown to be able to tolerate short-term increases in sodium chloride concentrations whereas algae and some invertebrates were considerably more susceptible. Cyanide and diuron at the levels detected in highway runoff are predicted to seriously inhibit algal growth which may have a knock-on effect on organisms higher up the food chain. Concentrations of glyphosate, aluminium, PAHs, oils, and ammonia that have been recorded in highway runoff are not expected to have an acute ecological effect even in the most severe runoff event with minimal dilution within the receiving water system. It is important to note that the 24 hour toxicity tests only determined the identified impact (e.g. mortality, effect, inhibition) and neither survival/recovery following exposure nor the effects of repeated exposures were investigated.

Short-term pollutant thresholds which integrate the different sensitivities of the range of tested species have been estimated either by extrapolation from the most sensitive test species or by using the Species Sensitivity Distribution (SSD) model. Crabtree et al. (2008) have used these results to derive Runoff Specific Thresholds (RSTs) for the protection of receiving water organisms from short term (6 hour and 24 hour) exposure to soluble pollutants in highway runoff. The 24 hour value is designed to protect against extreme exposure scenarios, whereas, the 6 hour value is designed to protect against more typical exposures resulting from highway runoff events. The 6 hour values are double those for 24 hour exposures which have been reported to be 40 µg/L, 21 µg/L and 60 µg/L for soluble fractions of Cd, Cu and Zn for water hardness lower than 50 mg/L and 1.2 µg/L and 1.3 µg/L for the two PAHs, fluoranthene and pyrene (Crabtree et al., 2009). Comparison with the event mean concentrations (EMCs) measured for 340 highway runoff events across the UK identified no exceedances for cadmium, fluoranthene and pyrene whereas a total of 21.7% and 20.5% exceedances were observed for the RST_{6h} values for copper and zinc, respectively. The percentage of failing events increased with AADT, particularly above 80,000, regardless of climate zone as shown in Table 3.2 for both RST_{6h} and RST_{24h} values. However, the RSTs are effectively discharge emission standards rather than receiving water standards and do not take into account the dilution which would occur in the receiving water. Nevertheless, these RST values have been incorporated into a design guidance tool (HAWRAT) developed by the UK Highways Agency (now Highways England) which provides highway designers and operators with information on where, and to what level treatment is required to manage the potential risk of ecological impact arising from highway runoff (DMRB, 2009; Gifford, 2008).

Table 3.2 Percentage of storm events failing RSTs according to road classification in terms of annual average daily traffic

AADT	Dissolved Cu RST_{24h} value (21 µg/L)	Dissolved Cu RST_{6h} value (42 µg/L)	Dissolved Zn RST_{24h} value (60 µg/L)	Dissolved Zn RST_{6h} value (120 µg/L)
5,000-14,999	40.0	0	10.0	0
15,000-29,999	43.3	13.3	31.7	10.0
30,000-49,999	41.2	15.7	35.3	15.7
50,000-79,999	51.8	5.7	45.8	13.3
80,000-119.999	58.2	21.9	47.3	20.9
120,000-200,000	80.4	54.3	86.9	56.5
Average exceedance	53.9	21.7	45.5	20.5

(Adapted from Crabtree et al., 2009)

The HAWRAT assessment tool also incorporates the results of a collaborative research project investigating the chronic effects of highway derived sediment-bound pollutants on the ecology of receiving waters (Gaskell et al., 2008). The scenarios under which contaminated sediment in runoff is likely to have a negative impact on receiving water ecology are identified. The results are used to develop Threshold Effects Levels (TELs) and Probable Effects Levels (PELs) for metal and PAH concentrations in sediment. The TEL is the concentration below which toxic effects are extremely rare. The PEL is the concentration above which toxic effects are observed on most occasions. Table 3.3 summarises the TELs and PELs derived from the study. In the absence of nationally agreed sediment guideline standards, the TELs and PELs have been agreed with the Environment Agency of England as a pragmatic approach reflecting current best practice and will be reviewed regularly and amended as necessary to reflect changing legislation or regulatory requirements.

Table 3.3 TELs and PELs for metal and PAH concentrations in sediment.

Sediment-bound pollutant	Units	TEL	PEL
Copper	mg/kg	35.7	197
Zinc	mg/kg	123	315
Cadmium	mg/kg	0.6	3.5
Total PAH	µg/kg	1,684	16,770
Pyrene	µg/kg	53	875
Fluoranthene	µg/kg	111	2,355

(from Gaskell et al., 2008)

Kayhanian et al. (2008) tested the toxicity of five different species to highway runoff collected as both grab and composite samples from three urbanized sites in Los Angeles, California. The tested species included three freshwater species (the water flea [*Ceriodaphnia dubia*], the fathead minnow [*Pimephales promelas*], and the green algae [*Pseudokirchneriella subcapitatum*]) and two marine species (the purple sea urchin [*Strongylocentrotus purpuratus*], and the luminescent bacteria [*Photobacterium phosphoreum*] using Microtox™). Toxicity results varied considerably throughout storm events for both freshwater and marine species but the first few samples were generally found to be most toxic with typically more than 40% of the toxicity being associated with the first 20% of discharged runoff volume and 90% of the toxicity occurring during the first 30% of storm duration. Typical results for *Ceriodaphnia dubia* are shown in Figure 3.1 which identifies the survival test response for runoff samples collected from two highway sites for the same storm event. The clear association of the toxicity with the first flush identifies the benefits of effectively treating the first portion of the stormwater runoff volume. Toxicity identification evaluation results indicated that copper and zinc were responsible for the toxicity in the majority of the samples (particularly to the water flea and fathead minnow) which is consistent with the findings of the comprehensive UK study (Hurle et al., 2006; Crabtree et al., 2009).

Bruen et al. (2006) have monitored the quality and quantity of run-off from 14 different two to four lane highway sites (AADT range: 2,513 – 50,7729) in Eire and investigated its impact on the composition and distribution of aquatic species (macroinvertebrates, fish and aquatic flora) in receiving watercourses possessing good to moderate ecological and chemical status. Contaminants found in the runoff waters include suspended solids, heavy metals (Cd, Cu, Pb and Zn), hydrocarbons including PAHs, chlorides, nitrates and phosphorus but not MTBE. Concentrations, although variable were consistent with those reported for similar site conditions in other European countries. Water quality upstream and downstream of highway runoff discharges showed little statistical difference between upstream and downstream locations but this was not mirrored by sediments for which metal levels downstream showed small concentration increases although only Cd, Cu and Zn exhibited statistically significant differences. Only Cd was detected at concentrations which exceed those at which probable effects could be expected.

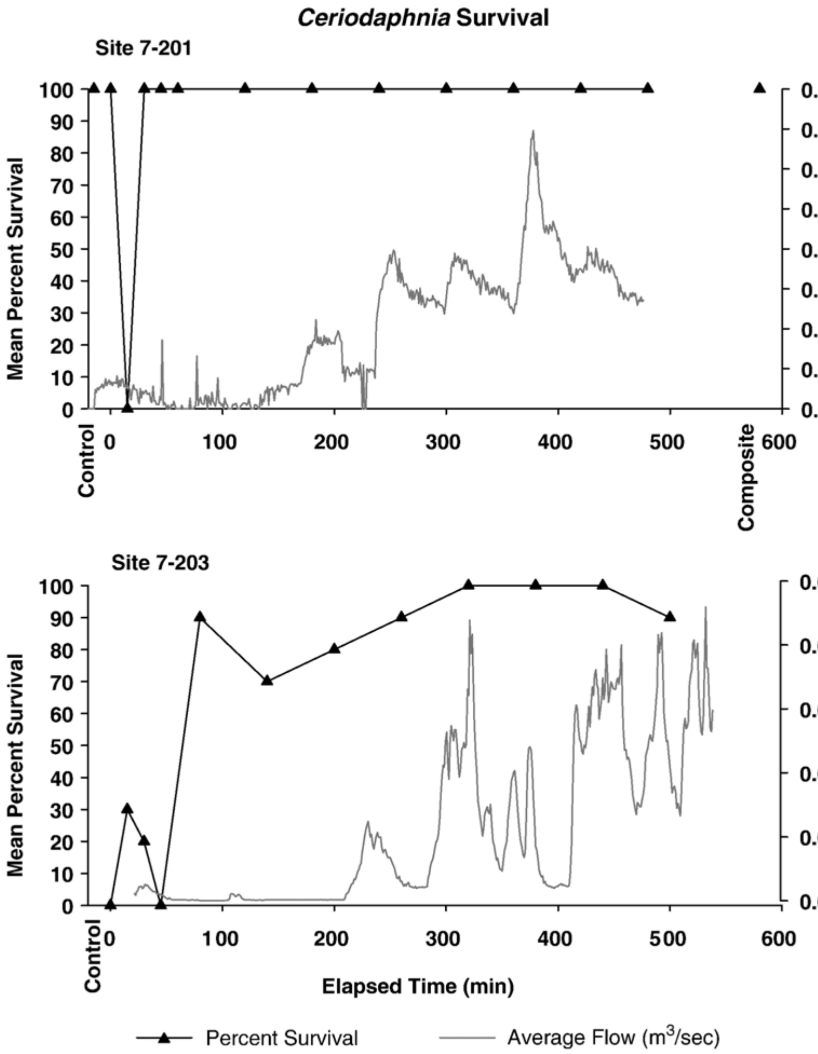


Figure 3.1 *Ceriodaphnia* survival test response to runoff samples from two highway sites during the same storm event (after Kayhanian et al., 2008)

None of the fish species (brown trout, *Salmo trutta*; stickleback, *Gasterosteus aculeatus*) demonstrated a negative impact due to highway runoff although this may have been masked at most sites by them being already impacted by upstream nutrient/organic pollution. Similarly, consideration of taxa numbers, individual numbers, percentage abundance and biotic indices showed no adverse effects could be detected in the macroinvertebrate fauna (family groups: Hirudinae; Crustacea; Mollusca; Oligochaeta; Ephemeroptera; Plecoptera; Hemipterans; Coleoptera; Trichoptera; Diptera), other than for Ephemeroptera species, where identified differences were related to limitations in the physical habitat or to impacts from other sources. In fact a decrease in Zn, Cu and Cd in *Gammarus* tissue downstream of some road runoff discharge points was found but explained by the development of an evolutionary tolerance to heavy metal pollution. Heavy metals were found in the tissue of vegetation (*Apium Nodiflorum;* Fool's water cress or European marshwort) near road drainage outfalls but away from the outfalls, no consistent pattern of statistically significant changes between vegetation upstream and downstream of the outfall was observed. However, relatively high levels of Zn were found to accumulate in Apium (watercress). Due to the rural nature of the highway sites, they were also impacted by other diffuse sources, particularly agricultural runoff making the separate identification of highway runoff impacts difficult.

Mayer et al. (2011) employed a battery of bioassays to assess the relationship between the toxicity and the chemical characteristics of highway runoff. Runoff samples (rain and snow melt) were collected from three sites in southern Ontario, Canada, representing different classes of multi-lane highways with different traffic intensities (high [AADT: 92,000]; intermediate [AADT: 31,100] and low [AADT: 15,460]). The species tested included the water flea (*Daphnia magna*), rainbow trout (*Oncorhynchus mykiss*), sub-mitchondrial particles and the Microtox test. Static non-renewal toxicity tests were used to screen samples of *D magna* for acute toxicity over 48 hour periods in undiluted runoff unless mortality rates were very high. A similar approach was used for rainbow trout (*Oncorhynchus mykiss*) but at a series of dilutions and for 96 hours. Sub-mitochondrial particle (beef heart bioparticles) assays, were used to detect conventional or reverse electron flow and hence the presence of bioavailable toxicants in some runoff samples. Runoff toxicity was also assessed using the standard Microtox™ test based on the reduction of bioluminescence of the marine bacterium *Vibrio fischeri*. In addition, estimates of sub-lethal toxicity (reproductive impairment) were determined using the water flea (*Ceriodaphnia dubia*) and rainbow trout were subjected to mixed function oxidase (MFO) tests. A selection of the results, expressed as IC25/IC50, LC50 and NOEC values, are shown in Table 3.4 for *D magna* and *C dubia* for several runoff events at one highway site. The data indicates that the *C. dubia* chronic test was more sensitive than the *D. magna* acute test when evaluating toxicity of runoff resulting from road salts. Generally, higher toxicities, and hence lower IC25, EC50 and NOEC values were observed with increasing conductivities of samples, corresponding to high concentrations of chloride. Moderate to strong acute and chronic toxicity responses were also generated by runoff samples containing elevated levels of Zn.

Surface runoff from the major multilane divided highway, with the highest traffic intensity (AADT: 92,000) demonstrated the highest acute and chronic toxicity on aquatic organisms in laboratory bioassays and also showed the most contamination (metals, PAHs and road salts). In general, a sharp decline in runoff toxicity through individual storms showed that the 'first flush' was the most toxic. The runoff samples containing high concentrations of road salts from winter maintenance were acutely toxic to *Daphnia magna*. Elevation of discharged metal loads due to corrosion of highway structures contributes to the toxicity of highway runoff. A strong MFO induction in rainbow trout has shown that this test is sensitive enough to determine the potential toxicity of PAHs in runoff. It is important to note that this study represents the 'worst case scenario', in which untreated highway runoff would directly enter a small receiving water body, with little dilution.

Table 3.4 Seasonal changes in runoff toxicity as indicated by *Daphnia magna* and *Ceriodaphnia dubia* toxicity tests at a highway site (after Mayer et al., 2011)

Date	Conductivity (µS/cm)	Cl mg/L	*C. dubia* IC25 (%)	IC50 (%)	NOEC (%)	*D. magna* EC50 (%)	Conditions
Dec 1	n/a		35	58	12.5	n/a	Rain
Feb 18	39,000	19,135	L*	L*	<3.12	12.5	Rapid snowmelt
Apr 7	23,000	10,960	4.5	7	1.75	20	Rain on snow
Apr 14	1,800	718	65	84	25	>100	Rain
Apr 17	5,000	2,410	47	58	25	>100	Rain
Aug 8	390	115	41	62	<25	n/a	Rain

The values in tables show dilutions of whole effluent obtained from interpolation when the effect occurred.
IC25 <40% – indicates severe acute effect.
IC25 <70% – indicates moderate acute effect.
IC50 >100 – indicates non-toxic sample.
L* – 48h median lethal concentration, LC50 4.5%.
IC25 – inhibiting concentration of whole stormwater effluent at which 25% lower rate of reproduction of *C. dubia* occurred.
IC50 – median effective concentration of whole stormwater effluent at which 50% lower reproduction of *C. dubia* occurred.
EC50 – median effective concentration of whole stormwater effluent at which 50 % of mortality in *D. magna* occurred.
NOEC – No Observed Effect Concentration.
n/a – no sample available.

Nevertheless, the data show that vehicular operation, road maintenance and metal highway structures are significant contributors to contaminant-associated toxicity in road runoff and there is a need for the careful planning and implementation of remediation strategies to mitigate the potential impacts of highway runoff pollution.

Chloride concentrations in highway runoff can be elevated following winter de-icing practices and therefore Elphick et al. (2011) carried out an evaluation of the toxicity of chloride to nine freshwater species. Acute and chronic toxicity tests were conducted using two species of water flea (*Ceriodaphnia dubia* and *Daphnia magna*), two species of worms (*Lumbriculus variegatus* and *Tubifix tubifex*), a chironomid (*Chironomus dilutus*), an amphipod crustacean (*Hyalella Azteca*), a rotifer (*Brachionus calyciflorus*) and two fish species (rainbow trout; *Onchorhynchus mykiss* and fathead minnow; *Pimephales promelas*). The results for the acute toxicity test are reported as the 24 hour to 96 hour median lethal concentrations (LC50s) with the chronic tests being based on 48 hour to 54 day inhibition concentrations for growth or reproduction. A water quality guideline for long term exposure to chloride of 307 mg/L was derived based on a 5th percentile value using the species sensitivity distribution approach. Water fleas were the most sensitive species tested and *Ceriodaphnia dubia* was used to evaluate if a relationship existed between sensitivity to chloride and water hardness Figure 3.2). The inference is that the guidelines may be overly conservative in waters with moderate to high hardness and may not be sufficiently protective under soft water conditions. Corsi et al. (2010) have also studied the toxicity of chloride to the water flea (*Ceriodaphnia dubia*). Thirteen Milwaukee, Wisconsin, USA streams receiving winter runoff of road salt were monitored and 54% found to exhibit toxicity. Chronic assay results indicated that no young were produced when chloride concentrations were 1770 mg/L or greater (43% of samples) and complete mortality was observed at chloride concentrations of 2420 mg/L and greater (38% of samples). Initial toxicity effects began between 600 and 1100 mg Cl/L.

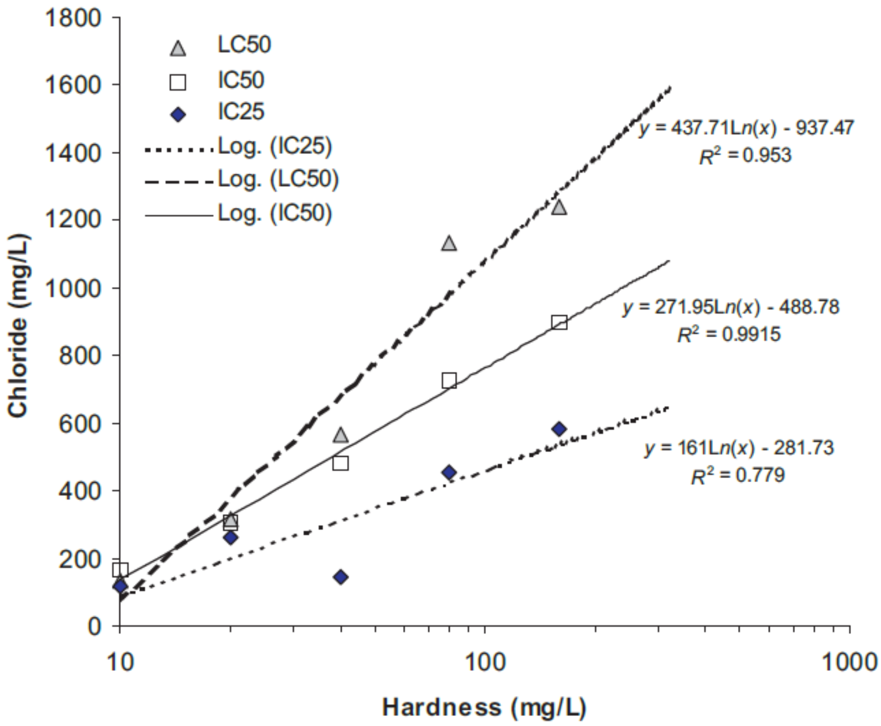


Figure 3.2 Relationship between hardness and sensitivity to chloride for reproduction (IC25 and IC50 inhibition concentrations) and survival median lethal concentrations (LC50s) (after Elphick et al., 2011).

3.1.2.2 Impacts on fish species

Fish are an important component of population and community assessments because they have long life spans (years to decades), are of interest to the public, are potentially economically valuable, are highly mobile, and so are indicative of the long-term health of a waterbody (Gilliom et al., 1995). Fish exist as a diverse group of species with different preferences for in-stream habitat. Fish populations can be characterised by the list of species present, fish size, fish abundance and condition, and by tissue analysis for the expected contaminants.

a) Brown trout (*Salmo trutta*)

The use of brown trout as part of a series of bioassays to assess toxicity have already been described in the context of the impact of highway derived soluble pollutants on the ecology of receiving watercourses (Hurle et al., 2006; Bruen et al., 2006) and for the assessment of the relationship between the toxicity and the chemical characteristics of highway runoff (Mayer et al., 2011). Using brown trout (*Salmo trutta*) as a sole model organism, Meland (2010) investigated the ecotoxicological effects of runoff from Norwegian highways and also man-made runoff from tunnel wash. Short term sub-lethal exposures (4 h) to traffic related contaminants caused a number of previously undetected molecular changes several hours after exposure. Several metals (including Al, Cu, Co, Fe, Pb and Sb) were accumulated in the gills initiating short term biological effects as signalled by biomarkers such as increased blood glucose levels, increased enzymatic activity of superoxide dismutase (SOD) and catalase (CAT) and increased concentrations of metallothionein (MT). The fish exposed to tunnel wash water demonstrated that PAHs and/or other organic contaminants were readily bioavailable, although normally strongly attached to particles. PAHs or other organic micropollutants were considered to be responsible for the expression of several classical genes (such as CYP1A1, CYP1B1, SULT and GST) indicating that oxidative stress was induced in the exposed fish (Meland et al., 2010a). The reduced growth in the sea trout population downstream of a sedimentation pond receiving tunnel wash water runoff indicated a long term negative biological effect with individuals typically being 21 % shorter than those from the upstream population. This finding questions the ability of sedimentation ponds to efficiently mitigate the environmental impacts of runoff derived from highways and tunnel washing activities.

Additional investigations of the impact of tunnel wash waters on a small stream following discharge from a sedimentation pond have been reported by Meland et al. (2010b). A range of contaminants including Cu, Pb, Zn, fluoranthene, pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(g,h,i)perylene and indeno(1,2,3-cd)pyrene were monitored at concerning levels. In situ size and charge fractionation techniques showed that although many of the contaminants were highly associated with particles and colloids (30%), a larger proportion (50%) occurred in the low molecular mass (LMM) fraction (<10 kDa) (Figure 3.3) which constitutes the more mobile and bioavailable species. This is consistent with the observed growth reduction in summer old brown trout in the lower parts of the stream compared to the upper parts.

b) Rainbow trout (*Oncorhynchus mykiss*)

In addition to the use of rainbow trout to determine the toxicity of chloride (Elphick et al, 2011) this species has also been used by Portele et al. (1982) to investigate the impacts of runoff from Washington State highways on aquatic ecosystems. High mortalities to rainbow trout fry were reported to be caused by elevated suspended solids concentrations.

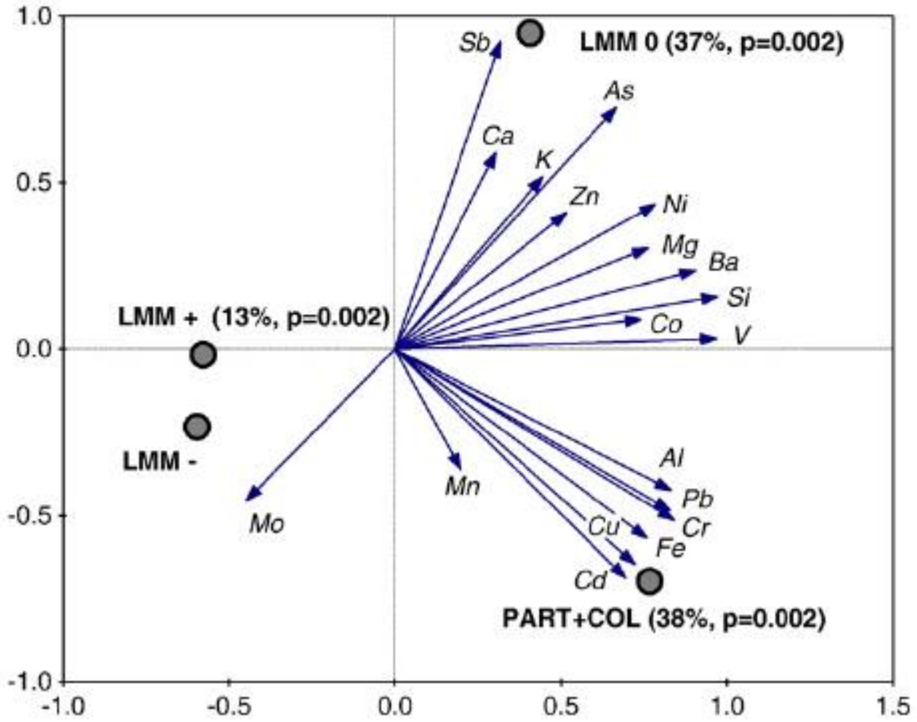


Figure 3.3 Multivariate redundancy analysis (RDA) plot showing the percentage association of metals with the particulate and colloid fractions and lower molecular mass fraction of waters discharged from a sedimentation pond.

c) Roach (*Rutilus rutilus*)
This species was studied by Hurle et al. (2006) as part of their comprehensive biological investigation of the toxicity of UK highway runoff. Toxicity test results for soluble copper indicated 24 hour LC50 values varying from 38.9 µg/L (at a water hardness of 23.0 mg/L) to 172.5 µg/L (at a water hardness of 234.7 mg/L) whereas the lowest toxicity of zinc (LC50; 3.1 mg/L at a water hardness of 23.0 mg/L) was not influenced by water hardness. Comparison with the maximum concentrations reported in highway runoff indicates that soluble Cu concentrations can exceed the measured LC50 values at all hardness values with zinc concentrations posing a potential threat to roach in soft waters. No mortality of roach was observed when exposed to the soluble fraction of 400 mg/L of total crankcase oil or 400 mg/L of total diesel oil. Based on an LC50 value of 0.45 g/L, roach were considered to be highly sensitive to potassium acetate toxicity.

d) Bullhead (*Cottus gobio*)
Hurle et al. (2006) found bullhead to be less sensitive than roach to soluble copper with determined 24 hour LC50 values ranging from 460.4 µg/L (at a water hardness of 12.0 mg/L) to 920.4 µg/L (at a water hardness of 213.3 mg/L). Increasing hardness also protected bullheads from zinc toxicity with 24 hour LC50s increasing from 3.12 mg/L (at a water hardness of 26.0 mg/L) to 9.92 mg/L (at a water hardness of 208.7 mg/L). Therefore, bullheads are unlikely to be susceptible to copper and zinc pollution in hard waters, even in undiluted highway runoff. A mortality level of 28.6 % was observed in bullheads exposed to 8.5 g/L potassium acetate indicating a potential toxicity in the most severe runoff events, although this would be compensated by a high of level dilution.

e) Common minnow (*Phoxinus phoxinus*)
The potential for copper and zinc in highway runoff waters to be toxic to the common minnow has been demonstrated by Hurle et al. (2006). As for other fish species soluble copper is more toxic than soluble zinc and for both metals there is a decrease in toxicity as the water hardness

increases. The most toxic reported 24 hour LC50 values are 188.7 µg/L for Cu (hardness 21.8 mg/L) and 1.77 mg/L for Zn (water hardness 25.3 mg/L) and these concentrations may be exceeded in some cases in undiluted highway runoff. Sodium chloride at a concentration of 6.5 g/L was not toxic to minnows over 24 hours and the 24 LC20 value of 4.64 mg/L derived for potassium acetate would be only marginally exceeded by the maximum recorded runoff concentration of 6.65 g/L.

The impact of road runoff on the common minnow has also been investigated by Grung et al. (2016) with the receiving water environment being represented by Norwegian sedimentation ponds. The objective of this study was to investigate if the minnows were affected by the highway runoff and to deduce if transfer of PAHs to the aquatic organisms occurred. Compared to a nearby river, the minnow from the sedimentation pond possessed higher levels of CYP1A enzyme and exhibited DNA damage, but both populations demonstrated high concentrations of PAH-metabolites in bile indicating that they had both been exposed to PAHs. Confirmation that fish health was being affected by PAHs in highway runoff was deduced by showing that minnow from a lake unaffected by traffic pollution had much lower levels of PAH-metabolites than the exposed fish, and were also in an improved condition.

f) Fathead minnow (*Pimephales promelas*)
As described earlier in this report, Kayhanian et al. (2008) have investigated the impact of highway runoff to fathead minnows and shown that approximately 50% of the estimated toxicity was associated with the first 20% of discharged runoff volume. In addition, TIE studies suggested that the primary cause of toxicity was due to the presence of cationic metals, specifically copper and zinc. In contrast, although the quality of bridge deck runoff in Nebraska, Canada was determined to be similar to that of highway runoff and frequently contained metals, 48-h acute toxicity assays using fathead minnows predicted that this would not have a long term impact on the dry weather water quality of the receiving stream (Swadener, 2014). Corsi et al. (2010) have used the same fish species to assess the toxicity caused to streams in Milwaukee, Wisconsin as a result of road salting during winter periods. Road salt contaminated runoff exhibited toxicity to fathead minnows in the majority of streams. In one stream, where the maximum chloride concentration reached 7730 mg/L, 72% of 37 samples exhibited toxicity in chronic bioassays and 43% in acute bioassays.

g) 3-Spined Stickleback *(Gasterosteus* aculeatus)
As described in Section 3.1.2.1, Hurle et al. (2006) and Bruen et al. (2006) have used the 3-spined stickleback together with 4 other fish species and one other fish species, respectively to assess the toxicity of highway runoff. In a field study (Bruen et al., 2006) failed to detect a toxic impact. However, 24-hour laboratory based toxicity tests using simulated highway runoff and receiving water pollutant concentrations indicated possible metal toxicity at low water hardness levels (19 mg/L) through determined LC50 values for soluble Cu and Zn of 239.0 µg/L and 2.68 mg/L (Hurle et al., 2006). Investigations of other highway runoff related pollutants suggested LC50 values for stickleback in excess of 13 µg/L for both pyrene and fluoranthene and no mortality was observed during exposure to NaCl concentrations of up to 2.1 g/L.

h) Perch (*Perca fluviatilis*)
The potential impact on perch (*Perca fluviatilis*) in a Norwegian lake receiving highway runoff (AADT; 29,600) has been assessed by comparison with the same species in a reference lake (Baekken, 1994). Although the highway derived pollutants resulted in a reduced diversity and abundance of the biotic communities in the contaminated lake, the only observed impact on perch were concentrations above background levels of lead in the liver and PAH in the flesh. With regard to the lead uptake, it is important to point out that this study was conducted prior to the use of lead-free petrol being fully implemented in Europe.

3.1.2.3 Impacts on Aquatic Invertebrates

Aquatic invertebrates represent a diverse group of taxa that live in, on, or near streambed sediments and include aquatic insects, molluscs, crustaceans and worms. Population and community assessments are widely applied as aquatic invertebrates possess life spans of months to years, live in close association with streambed sediments, are relatively sedentary and good indicators of local water quality. Assessment of aquatic invertebrate populations can involve investigation of the numbers and types of taxa present within a predefined study area as they exhibit different tolerances to contaminants that are found in sediments and the water column. Some genera are highly sensitive and the presence of low pollutant concentrations can result in their elimination from the benthic community. In contrast, other genera are more tolerant of pollution and would only be influenced in terms of reduction of numbers at high contaminant levels. Thus, measures of the presence, absence, and abundance of various taxa can be used as an indicator of aquatic pollution. A number of different biotic indices (or scoring systems) have been developed for assessing aquatic habitat quality based on the variety and numbers of aquatic invertebrate species present (Metcalfe, 1989). Most of these were specifically developed to establish responses to point discharges of organic effluents.

Examples of existing scoring systems developed in the UK include the biological monitoring working party (BMWP), the average score per taxon (ASPT), the River Invertebrate Prediction and Classification System (RIVPACS) and the EPT richness index. The BMWP biotic scoring system (Biological Monitoring Working Party,1978) assigns different scores to different families of invertebrates with higher values being allocated to the most sensitive species. The cumulative score represents the BMWP value. A simplified version of the original and revised BMWP scoring systems, in which the family group sub-divisions have been omitted, is shown in Table 3.5. The revised version is based on the analysis of frequency of occurrence of the families recorded in approximately 17,000 samples (Paisley et al., 2014).The ASPT value is derived from the community BMWP score by dividing by the number of scoring taxa represented and has a value between 1 and 10 with the lower values representing the poorest water quality conditions and vice versa (Armitage et al., 1983). RIVPACS is based on the same principles as the BMWP system and represents an aquatic biomonitoring system for assessing water quality in freshwater rivers (Wright et al., 2000).The EPT richness indicator estimates water quality by the relative abundance of three major orders of stream insects (Ephemeroptera, Plecoptera, Tricoptera) that have a low tolerance to water pollution. A large percentage of EPT taxa compared to the total taxa present indicate a high water quality.

Table 3.5 Ranges of BMWP scores for aquatic macroinvertebrates identified to common name only (i.e. not including subdivisions according to family name).

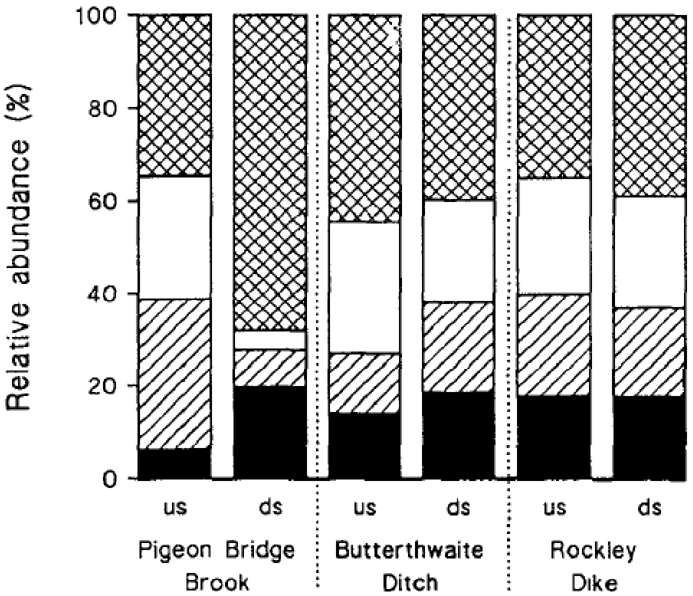
Common Name	Original BMWP Score	Revised BMWP Score
Snails	3-6	1.8-7.5
Limpets and mussels	3-6	3.6-5.6
Worms	1	3.5
Leeches	3-4	0-5.0
Crustaceans	3-8	2.1-9.0
Mayflies	4-10	5.3-11.0
Stoneflies	7-10	9.1-12.5
Damselflies	6-8	3.5-6.4
Dragonflies	8	5.0-8.6
Bugs	5-10	3.7-8.9
Beetles	5	2.6-7.8
Alderflies	4	4.5
Caddisflies	5-10	6.6-10.9
True flies	2-5	3.7-5.8

Bruen et al. (2006) have studied the hydrobiology, both upstream and downstream of highway discharges, at 14 stream sites in Ireland. The mean BMWP scores ranged from 40 to 148 and at nine of the sites higher BMWP scores were recorded at the upstream locations although none were significantly different ($P>0.05$) to those downstream. The mean ASPT scores ranged from 4.1 to greater than 7 with some differences being evident between upstream and downstream sites however only four sites demonstrated significant differences. The mean % EPT abundances ranged from a low of 3% up to 76% with higher values being observed at 13 of the upstream sites but again there were no significant differences between upstream and downstream locations. Overall, there was no concrete evidence that highway road runoff was having a negative impact on the macroinvertebrate communities in the downstream sections of the streams examined. In addition, the application of a two way indicator species analysis (TWINSPAN) clustering technique to the collected data showed a low % heterogeneity in species composition and abundance confirming no significant differences between upstream and downstream sites. The conclusion reached was that that no adverse effects from highway drainage could be discerned from the behaviour of macroinvertebrate communities in the investigated streams.

An earlier investigation in Scotland (McNeil and Olley, 1998) involved aquatic invertebrate monitoring upstream and downstream of highway discharges into 5 streams selected because of the absence of any interfering factors in the vicinity of the discharges. Recorded BMWP and ASPT values obtained from 12 sampling occasions varied from 22 to 147 and 3.4 to 6.5, respectively with the ASPT parameter appearing to be more sensitive in identifying decreases from upstream to downstream locations. However, there was no evidence that highway discharges, during normal operating conditions, were impacting on the biology of any of the monitored receiving watercourses. One of the reasonably intolerant species (G pulex; original BMWP score of 6; revised BMWP score 4.5 [Paisley et al., 2014]) was found at every monitored site, and there was no evidence that it had been affected by solids or any associated toxins. In contrast the macroinvertebrate diversity was clearly found to be suppressed during the highway construction phase probably due to siltation effects on the stream beds.

The effects of motorway runoff on the water quality, sediment quality, and biota of seven small streams in northern England were investigated over a 12-month period by Maltby et al., (1995a). Increases in the sediment concentrations of total hydrocarbons, aromatic hydrocarbons, and heavy metals and in the water concentrations of heavy metals and selected anions were noted 100m downstream of motorway runoff discharges. The dominant PAHs in contaminated sediment were phenanthrene, pyrene and fluoranthene, whereas the dominant metals were zinc, cadmium, chromium, and lead. Between the stations upstream (440 m above outfall) and downstream of discharges, changes in the diversity and composition of the macroinvertebrate assemblages were determined using BMWP and ASPT scores and additionally the log series diversity index (α) (Fisher et al., 1943) and a modified Sorenson index (C_N) (Magurran, 1988) were applied. The diversity (α values) of the macroinvertebrate communities was reduced at four out of the seven monitored stations receiving highway runoff where lower biotic scores were also recorded indicating that contaminated downstream stations had fewer sensitive species than did the less polluted upstream sites. Thus, stoneflies (Plecoptera), gammarids (Amphipods), caddisflies (Trichoptera) and snails (Molluscs) tended to predominate upstream of highway discharges with chironomid larvae (Diptera) and tubificid worms (Oligochaeta) being more abundant downstream. Reductions in macroinvertebrate diversity were associated with reductions in the processing of leaf litter and a change from an assemblage based on benthic algae and coarse particulate organic matter to one dependent upon fine particulate organic matter and dominated by collectors as opposed to shredders (Figure 3.4). The changes in macroinvertebrate distributions have been tentatively linked to toxic effects as no significant between-station differences were noted in either the abundance of epilithic algae or detritus and associated fungi. The increased abundance of chironomids and oligochaetes and the decreased abundance of *G pulex* could not be explained by changes in substrate particle size or total organic carbon content.

BMWP and ASPT scores as well as Shannon diversity index (Hughes, 1978) values have been compared for aquatic invertebrates collected from upstream and downstream sites (relative to highway discharges from the A12 and A14 trunk roads) in nine rivers located in East Anglia, UK (Perdikaki and Mason, 1999). Both roads are two-lane dual carriageways with AADTs of 67,000 (A12) and 31,557 (A14). Samples were only collected in the spring and summer seasons. The BMWP scores (range: 48-165) were higher at upstream sites on 70% of the sampling occasions for those rivers crossed by the A12 but the ASPT scores (range: 3.82-5.16) and the Shannon diversity indices (range: 1.53-2.63) were higher for upstream sites on only about half of the occasions. The indices were higher in upstream sites on half of the occasions for the rivers crossed by the A14. The only significant difference ($p < 0.05$) was for higher BMWP scores at upstream sites in streams crossed by the A12 during the summer survey suggesting that road runoff might have an impact on these rivers during low flow conditions. Although additional subtle effects, not detected by the biotic indices, may have been present the overall results indicate that there were no major impacts on the macroinvertebrate community due to road runoff at the monitored sites.



Solid bars + predators; hatched bars = shredders; open bars = scrapers; cross-hatched bars = collectors

Figure 3.4 Mean relative abundance of functional feeding groups in three streams above (us) and below (ds) motorway runoff outfalls (after Maltby et al., 1995a)

Chen et al. (2009) have assessed the impact of highway construction on the biotic health of nine streams in West Virginia, USA. Ecological conditions were characterised using the West Virginia stream condition index (WVSCI), which incorporated six normalized metrics based on family level data. The six metrics were EPT (Ephemeroptera, Plecoptera, Trichoptera) taxa, total taxa, % EPT, % Chironomidae, % top 2 dominant taxa, and the Hilsenhoff Family Biotic Index (HBI). Statistical analysis using paired t tests found that differences between WVSCI score at upstream and downstream sites were not significant both before and during construction but were significant after highway construction as the result of an increasing representation by chironomids whilst the numbers of Ephemeroptera, Plectoptera, and Trichoptera reduced. However, the overall good biological condition remained unchanged.

3.1.2.3.1 Comparisons of different species of macroinvertebrates

a) Freshwater shrimp (Gammarus pulex), Water hoglouse (Asellus aquaticus) and Alderfly (*Sialis lutaria*)

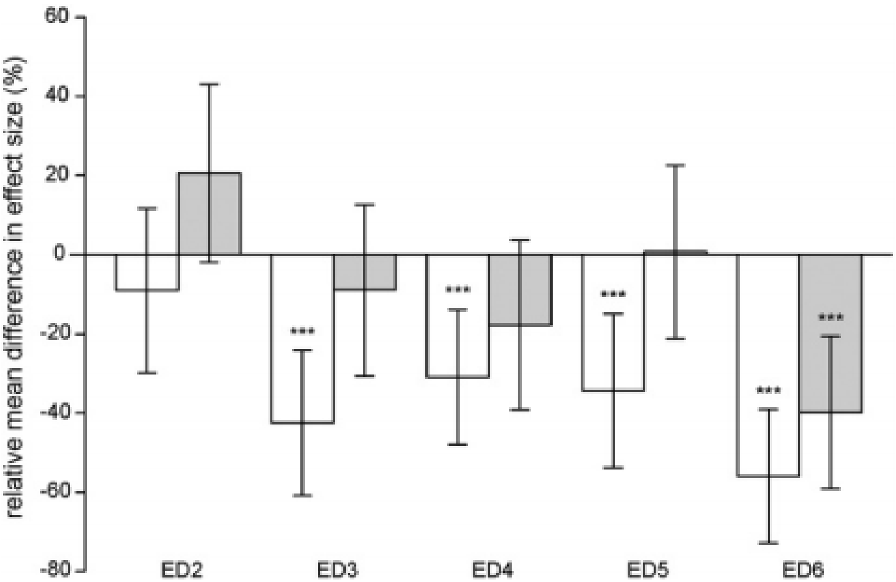
The susceptibilities of these macroinvertebrates to metal pollution in highway runoff have been investigated in animals collected from streams influenced by ten road crossings (two-lane dual carriageways) in Eastern England (Perdikaki and Mason, 1999). At each site, sediment and macroinvertebrate samples were taken from four to six locations and subjected to total metal (Zn, Pb and Cd) extractions. The only metal showing significant relationships between sediment and invertebrate concentrations was Pb whereas the Cd concentration in *Gammarus* was significantly negatively correlated with Cd in sediment. No significant differences were observed between upstream and downstream sites with regard to metal concentrations in both invertebrates and sediments indicating no contamination to either sediment or fauna from highway runoff at the monitored downstream sites.

b) Various macroinvertebrates including gammarids etc

A number of studies have investigated the existence of sodium chloride induced macroinvertebrate drift behaviour and/or mortality and found this to be variable among different taxa and dependent on concentration and exposure time (Chadwick, 1997; Goetsch and Palmer, 1997; Kundman, 1998; Williams et al., 2000). To provide further information on this behaviour, Blasius and Merritt (2002) have conducted field investigations in two Michigan, USA streams receiving runoff from major highways to examine the effects of residual road salt on stream macroinvertebrates and supported this with controlled laboratory experiments. The macroinvertebrates were chosen to represent a number of different trophic levels, habitat requirements, respiratory systems, and phenology and included gammarids, mayflies, stoneflies and caddisflies. Field studies investigated leaf litter processing rates and functional feeding group composition within the receiving streams revealing that leaves were processed faster at upstream sites than at locations downstream from road salt point source inputs. However, the main impact on macroinvertebrate activity was sediment loading resulting in partial or complete burial of leaf packs and confounding normal leaf pack colonization. No significant differences in the diversity and composition of invertebrate functional feeding groups could be attributed to road salt between upstream and downstream locations. Laboratory studies determined the effects of increasing NaCl concentrations on aquatic invertebrate drift, behaviour and survival. Laboratory drift and acute exposure experiments demonstrated that drift of gammarids may be affected by NaCl concentrations greater than 5000 mgL for a 24-h period. This amphipod and two species of caddisflies exhibited a dose response to salt treatments with 96-h LC50 values of 7700 and 3526 mg/L, respectively. Most other invertebrate species and individuals were unaffected by NaCl concentrations up to 10,000 mg/L for 24 and 96 h, respectively.

A field-based microcosm approach has been used to determine whether sediments that receive highway runoff are toxic to indigenous aquatic macroinvertebrates (mosquitoes, midges and moths) present in the Greater Melbourne Area, Australia (Pettigrove et al., 2007). Sediments collected from areas draining 3 three lane highways (AADT; 100,000 – 170,000) were placed in 20 L microcosms along the littoral zone of a non-polluted wetland from where aquatic insects are able to emerge to randomly lay eggs in the microcosms. Based on measurements of occurrence and abundance, the microcosm sediments were shown to exhibit different toxicities to several taxa. The abundance of the mosquitoes (*Paratanytarsus grimmii*, *Polypedilum leei* and *Oxyethira Columba*) significantly increased with increased concentrations of contaminants in the sediments, and appeared to be most influenced by nutrient enrichment. The occurrence of the aquatic insect (*Tanytarsus fuscithorax*) significantly declined with increased concentrations of zinc in surface waters due to leaching from sediments. The abundance of the midge (*Cricotopus albitarsis*) was significantly higher in nutrient-enriched sediments, but significantly declined in high zinc concentrations in surface waters. There were significant negative correlations between the occurrence of the aquatic insects (*Larsia albiceps*, *T. fuscithorax* and *Procladius* spp.) and copper or total petroleum hydrocarbon (TPH) concentrations in sediments. The application of field based microcosm studies demonstrates the different behaviours of indigenous macroinvertebrates when exposed to sediments contaminated with Cu, Zn and TPH due to highway runoff.

As part of a study on the impacts of successive land uses on the ecosystem processes in a 5 km stretch of a second-order stream in southern Germany, Englert et al. (2015) have investigated the role of highway runoff (AADT: 50,000) during a winter and summer season. Significant shifts in the macroinvertebrate community composition, which coincided with substantial impairments (up to 100%) in the macroinvertebrate-mediated leaf decomposition were observed either side of the highway discharge point. The main driver has been identified as alterations in water quality as a result of highway discharges rather than morphological modifications. Decreases in the abundances of gammarids and mayflies were matched by increases in chironomids and tubifex species below highway discharges. Gammarid feeding rates, leaf litter quality and shredder abundance are known to be prime ecological impacts on aquatic invertebrate communities impacted by highway runoff (Figure 3.5). This may be partly mediated by increased microbial activity (leading to the higher leaf decomposition) at the downstream site).



Asterisks denote significant differences, p< 0.001 (***).

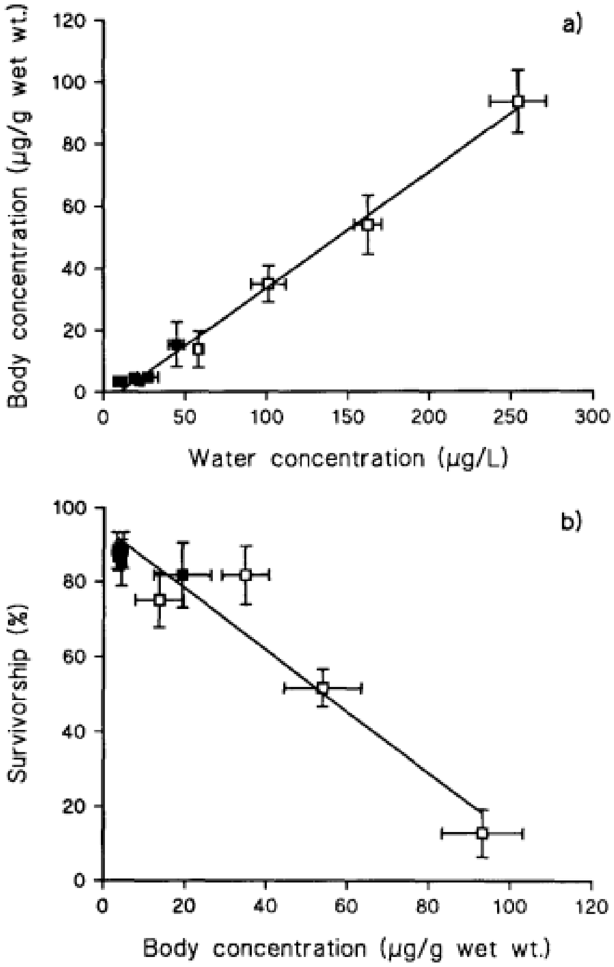
Figure 3.5 Relative mean difference (with 95% CI) in gammarid feeding rate obtained by a fixed-effect meta-analysis of in situ bioassay data between the upstream (ED1) and each downstream site during winter (white bars) and summer (gray bars). (After Englert et al., 2015)

Meland et al. (2013) have investigated the concentrations of several metals (Al, As, Cd, Co, Cr, Cu, Fe, Ni, Pb, Sb and Zn) in water, sediment and aquatic insects (dragonfly, mayfly, damselfly larvae) obtained from 5 wet sedimentation ponds (four receiving highway runoff [AADT: 37,000 to 53,000]) and one receiving tunnel washoff from 3 tunnels). Water and sediment samples from the sedimentation ponds were only moderately contaminated and not very distinct from two small ponds unaffected by traffic. However, the pond receiving tunnel wash water was contaminated by high levels of Cu and Zn both in water and sediment, and to some extent Ni and Pb in water. The metal concentrations in sediment and water were only marginally correlated with the metal body burdens of the sampled invertebrates.

3.1.2.3.2 Individual Macroinvertebrates

a) Freshwater shrimp; *Gammarus pulex*

Maltby et al. (1995b) have focussed on the benthic amphipod *Gammarus pulex* in an attempt to establish the existence of a causal relationship between the previously reported changes in water/sediment quality and biology in streams receiving highway runoff from a major motorway (Maltby et al., 1995b). The abundance of this species was considerably reduced downstream of the discharge point. However, toxicity tests using stream water contaminated with motorway runoff was not toxic to *G. pulex* but exposure to contaminated sediments resulted in a small but statistically significant reduction (10%) in survival over a period of 14 days. Sediment manipulation experiments identified hydrocarbons, copper, and zinc as potential toxicants and spiking experiments confirmed the importance of hydrocarbons. Further investigation using fractionation studies identified most of the observed toxicity as being due to the fraction containing polycyclic aromatic hydrocarbons as opposed to metals. Exposure of gammarids to contaminated sediments and water spiked with sediment extract resulted in aromatic hydrocarbons being accumulated in direct proportion to exposure concentrations (Figure 3.6). The combination of field observations and laboratory toxicity experiments identifies hydrocarbons, and particularly PAHs, as contributing to the toxicity of Gammarus pulex and as possibly being responsible for the most serious ecological implications.



Open symbols represent animals exposed to downstream sediment extract; solid symbols represent animals exposed to upstream sediment extract. Error bars are 1 SE. Lines fitted by regression analysis.

Figure 3.6 Relationship between (a) the concentration of aromatic hydrocarbons in the test solution and their accumulation by *G. pulex* and (b) survival and whole-body aromatic hydrocarbon concentration (after Mattby et al., 1995b)

Further investigations, using *G pulex* as the indicator organism, were carried out to determine whether PAHs were the major toxicants in sediment extracts (Boxall and Maltby,1997). By performing a series of toxicity tests with PAH mixtures, the toxic fraction of an extract of runoff-contaminated sediment, and a whole sediment extract, three PAHs (pyrene, fluoranthene, and phenanthrene) were shown to be responsible for the toxicity of a sediment extract. The possibility of spatial or temporal variations in the major toxicants was also investigated by performing tests on sediment extracts obtained from a number of sites at different times. This confirmed that six PAHs (anthracene, phenanthrene, fluoranthene, pyrene, chrysene, benzo(a)anthracene) accounted for the toxicity of the toxic fraction and three of these PAHs (phenanthrene, fluoranthene, and pyrene) accounted for 30.8 to 120% of an extract's toxicity. The relative toxicities of these three PAHs, as defined by 14 day LC50 values, are compared in Table 3.6. Possible reasons for non-explanation of all the observed toxicity include the presence of additional unidentified toxicants in the extracts, some form of interaction between the PAHs and other components of the extract, and/or metabolism of the PAHs to more toxic compounds. Individual consideration of the PAHs showed that pyrene accounted for most of the toxicity (44.9%), followed by fluoranthene (16%) and phenanthrene (3.5%).

Table 3.6 *Gammarus pulex* 14 day LC50 (μg/L) values together with 95% confidence limits values for phenanthrene, pyrene and fluoranthene (Boxall and Maltby, 1997)

PAH	LC50 (μg/L)	95% Confidence Limits	r^2	n
Phenanthrene	300.9	50-550	0.97	5
Pyrene	27.1	24-31	0.97	6
Fluoranthene	95.8	81-111	0.97	6

b) Estuarine gammarid (Grandidierella japonica)

This species, which is native to the Northwest Pacific, has been utilised in a Japanese study (Hiki and Nakajima, 2015) to investigate the influence of salinity (over a salinity gradient from 5 to 35‰) on the release of toxicity from road dust (contaminated with metals and PAHs) to highway runoff. Increasing the salinity consistently resulted in increased mortalities of the amphipod after 10 days of exposure as well as a decrease in short-term microbead ingestion activity. Assuming that microbead ingestion represents a proxy for feeding activity, the high mortality at 35‰ salinity can be attributed to aquatic exposure as opposed to dietary exposure. The results suggest that contaminated road dust may have a considerable impact on benthic organisms at high salinity levels.

c) Mussel shrimp (*Heterocypris incongruens*)

The toxicity of road dust to benthic organisms has been investigated using the ostracod, *Heterocypris incongruens,* through 6 day direct exposure experiments to road dust/water mixtures (Watanabe et al., 2011). Road dust collected from 6 heavy traffic areas (AADT: 17,530 – 64,715) caused high mortality of the ostracod whereas road dust from a residential area showed no toxicity (Figure 3.7). Interestingly, wet road dust that had been separated from a road dust water mixture after a holding time of 1 hour or 24 hours did not exhibit lethal toxicity, whereas the water soluble fraction of the mixture did cause high mortality of the ostracod. However, after conditioning the road-dust water mixture for 7 days both the wet road dust and the water soluble fraction showed lethal toxicity. The inference is that contact between road dust and rain water results in a water soluble fraction containing the majority of the toxicants. However, over prolonged incubation, such as could occur in sediment retained in a drainage system, the particle bound fraction again becomes toxic to the ostracod. This may be due to re-absorption of toxic compounds from water to the solid phases by equilibration or as a result of colloidal aggregation.

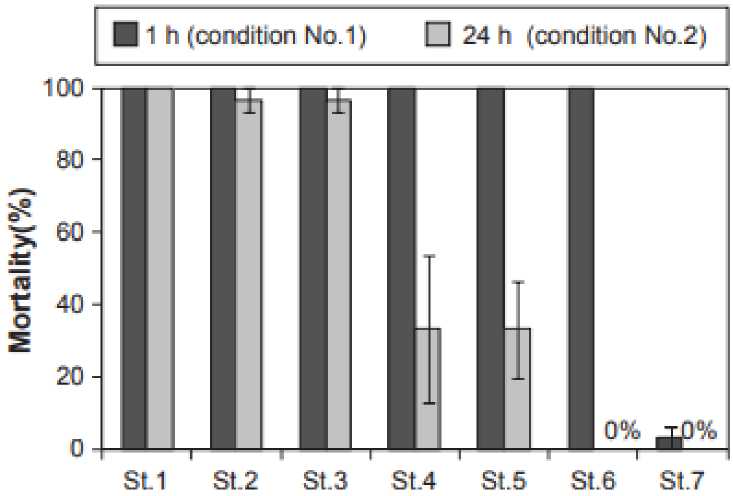


Figure 3.7 Results (mean± S.E) of toxicity tests on the whole road dusts collected at seven sampling stations (ST.7=residential site) (after Watanabe et al., 2011)

3.1.2.4 Impacts on algae, bacteria and fungi

There have been a limited number of biomonitoring studies which uniquely involve the use of algae, bacteria or fungi to assess the impact of highway runoff on receiving water. However, in a study indirectly related to road runoff, Huber et al. (2001) investigated the toxicity to algae (*S capricornutum*) of the synthetic leachate from 30 road construction and repair materials (e.g asphalt, Portland cement, plasticisers) used in the USA. The preparation of the leachates by shaking with deionised water for 24 hours represented a more extreme exposure than would be posed by rainfall running directly off the material surface. The highest toxicity to *S capricornutum* was exhibited by ammoniacal copper zinc arsenate (a wood preservative; EC50 value 0.9%), methymethacrylate (a sealant; EC50 value 2.5%) and municipal solid waste incinerator bottom ash (EC50 value 3.0%). The elevated toxicity of ammoniacal copper zinc arsenate was identified as being due to the constituent metals. Other metals found in the tested materials and identified as being responsible for detected toxic effects were Al, Pb and Hg. Comparable toxicity tests using *Daphnia magna* generally indicated that road materials were non-toxic effect except for a crumb rubber/asphalt concrete composite which demonstrated a LC50 value of 44% with the potential toxicants being identified as benzothiazole, Al and Hg.

Two algal species representative of UK watercourses (*Selenastrum capricornutum* and *Synedra delicatissima*) have been exposed to key pollutants in highway runoff using laboratory based toxicity tests (Hurle et al., 2006). For *Selenastrum capricornutum*, IC50 values for Cu, Zn and Cd of 194.9 µg/L, 73 µg/L and 125 µg/L, respectively were obtained under low hardness conditions (< 25 mg/L). Comparison of these values with typical highway runoff concentrations for these metals, indicates that Cu and particularly Zn could pose a problem under conditions of low dilution. *Synedra delicatissima* was observed to be considerable more sensitive to sodium chloride (IC50; 0.74 g/L) compared to *Selenastrum capricornutum* (IC50; 11.56g/L) suggesting that runoff induced by heavy rainfall immediately following salt application to a highway could have a serious impact on this algal species. LC50 concentrations for the green alga *S. capricornutum* exceed 30 µg/L pyrene and 30 µg/L fluoranthene as growth was not inhibited by these concentrations. Similarly, no impact was observed when *S. capricornutum* was exposed to either 400 mg/L diesel or 400 mg/L crank case oil. The herbicides diuron and glyphosate responded differently to the algal species. Although diuron demonstrated toxicity to both species, the growth of *Selenastrum capricornutum* was inhibited by 86.1 % at a concentration of 3 µg/L whereas *Synedra delicatissima* showed a 63.1 % inhibition of photosynthesis when exposed to 18.8 µg/L diuron.

In contrast, glyphosate is not expected to impact on algae in receiving watercourses based on a measured 24 hour IC20 value of 3.44 mg/L for *S. delicatissima.*

As previously reported (Section 3.1.2.1), Kayhanian et al. (2008) have utilised a series of toxicity tests to assess the impacts of runoff samples collected from Los Angeles highways. Included in these tests were the 96 hour growth inhibition of the freshwater species unicellular green algae (*Pseudokirchneriella subcapitatum*; also known as *Selenastrum capricornutum*) and the luminescent bacteria (*Photobacterium phosphoreum* also known as *Vibrio fischeri*) using the Microtox test. Unfortunately, both the Microtox and green algae tests were found to have limited utility due to enhancement of luminescence or cell growth in many samples as a consequence of the availability and concentration of nutrients in the runoff, potentially confounding the identification of toxicity patterns. However, when discernible the greatest toxic response of the green algae usually occurred in samples collected during the first 60 minutes of discharge. In the case of the Microtox test, typically only the first one or two grab samples were toxic at the selected test concentration of 9.4%.

Maltby et al. (1995a) have surveyed algae and fungi assemblages upstream and downstream of highway runoff discharges at 3 stream sites in northern England, UK. Representatives of 20 algal genera were recorded and although the algal assemblages at downstream locations were more diverse and contained more genera, none of these differences were statistically significant. The diversity of the aquatic hyphomycete population was only affected at the most impacted site. As there were no significant between-station differences in either the abundance of epilithic algae or fungi, it was concluded that the observed changes in macroinvertebrate distributions were due to direct toxic effects as opposed to the existence of feeding limitations (see also Section 3.1.2.3).

As one component of a battery of biotests (see Section 3.1.2.1), Mayer et al. (2011) employed the Microtox test to both the soluble and solid phases of runoff collected from a major multi-lane highway in Ontario, Canada. The soluble phase test was performed on samples concentrated 10 times by flash evaporation to compensate for dilution required during the test procedure and the Microtox™ Solid Phase test was performed on the same equipment as the liquid Microtox acute test. The results of the Microtox™ sediment tests indicated an increasing toxicity over time (Figure 3.8) which is the opposite trend to that displayed by the toxicity of the aqueous portion of runoff, where a rapidly diminishing toxicity with time is observed due to a decreasing concentration of soluble toxicants.

The impacts of roadway runoff, from an interstate highway in Konoxsville, Tennessee, USA, on the microbiological quality of receiving streams have been monitored at points above, below and at the point of discharge (Wyckoff et al., 2017). The microbiological quality was assessed during storm events with both cultivation-dependent fecal bacteria enumeration and cultivation-independent high throughput sequencing techniques. Enumeration of total coliforms (as a measure of fecal microbial pollution) found consistently lower total coliform counts in highway runoff than those in the stream water indicating, as expected, that roadway runoff was not a major contributor of microbial pollutants to the receiving stream. Further characterization of the microbial community in the stormwater samples by 16S ribosomal RNA gene-based high-throughput amplicon sequencing revealed significant differences in the microbial composition of highway runoff and the receiving stream. These differences demonstrate that road runoff does not have a major influence on the stream in terms of microbiological quality. The results from both fecal bacteria enumeration and high throughput amplicon sequencing techniques were consistent in confirming that highway runoff is not a primary contributor of microbial loading to a receiving stream.

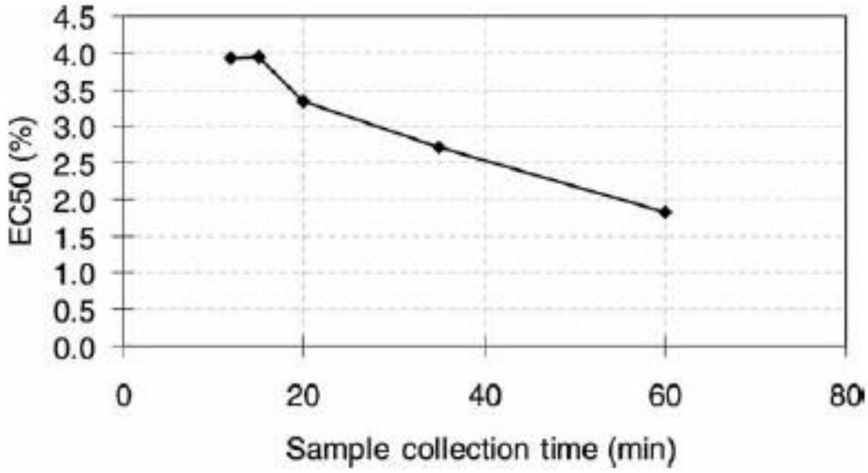


Figure 3.8 Decreasing trend in the toxicity of runoff solids over the course of a storm event as monitored using the Solid Phase Microtox™ Test

3.1.2.5 Impacts on aquatic plants

In an investigation which mainly concentrated on the impact of highway runoff derived PAHs to fish in sedimentation ponds (see Section 3.1.2.2 e), Grung et al. 2016 also compared the behaviour of aquatic plants and algae in 2 sedimentation ponds with a reference pond not receiving highway runoff. Periphyton was collected from the pond surface, pondweed (*Elodea Canadensis*) was sampled from the bottom, and the broad leafed pondweed (*Potamogeton natans*) was extracted via the roots. The PAH concentrations in the different plants varied considerably between sites and species and measured levels of the sum of 16 EPA PAHs (210 to 48,000 ng^{-1} g ww) were higher than in adjacent sediments and may be a better indicator of environmental risk. The plants preferentially accumulated the high molecular PAHs, both from sedimentation ponds with a petrogenic PAH isomer ratio in the sediments and from a lake with a pyrogenic PAH isomer ratio in the sediments.

Fool's water cress or European marshwort (*Apium nodiflorum*) is commonly found in Irish rivers and has been used by Bruen et al. (2006) to monitor the ability of the shoots and roots to accumulate metals (Cd, Cu, Pb and Zn) from water and/or sediment adjacent to highway runoff outfalls. Vegetation samples were collected from 2 rivers receiving runoff from the N7 highway at locations 50 m upstream, within 10 m and more than 80 m downstream of the runoff outflow pipe. Zinc was found at considerably higher concentrations than the other three metals indicating the ability of *Apium nodiflorum* to accumulate it. Higher metal levels were consistently found in the roots compared to the shoots but no consistent pattern of statistically significant changes between vegetation upstream and downstream of the outfall was observed.

3.1.2.6 Impacts on amphibians

Two research projects from Norway have investigated the behaviours of the developmental stages of the common frog (*Rana temporaria*) collected from sedimentation ponds receiving motorway runoff. Meland et al. (2013) focussed on the temporal accumulation (48 days) in both eggs and tadpoles. Other than for Zn, the concentrations of metals (As, Cd, Co, Cr, Cu, Ni, Pb, Sb and Zn) in both developmental stages appeared to increase during the studied period (Figure 3.9a). A principal component analysis indicated the existence of a strong linear metal gradient and a subsequent one-way ANOVA revealed a significant time-dependent accumulation (r^2 = 0.8, p = 0.0001) (Figure 3.9b). The steepest increase was observed between the developmental stages of egg and tadpole up to day 6 and day 27, respectively which may reflect fundamental physiological differences between the two developmental stages. During the 'egg-stage' metals are only able to accumulate passively whereas in the ‘tadpole-stage’ both active and passive uptake of metals occurs through the ingestion of food, respiration through gills and passive diffusion through the body surface. A subsequent levelling off of the accumulation rate for most of the metals after day 27 suggests that the metal body burdens have achieved a threshold level under the prevailing conditions.

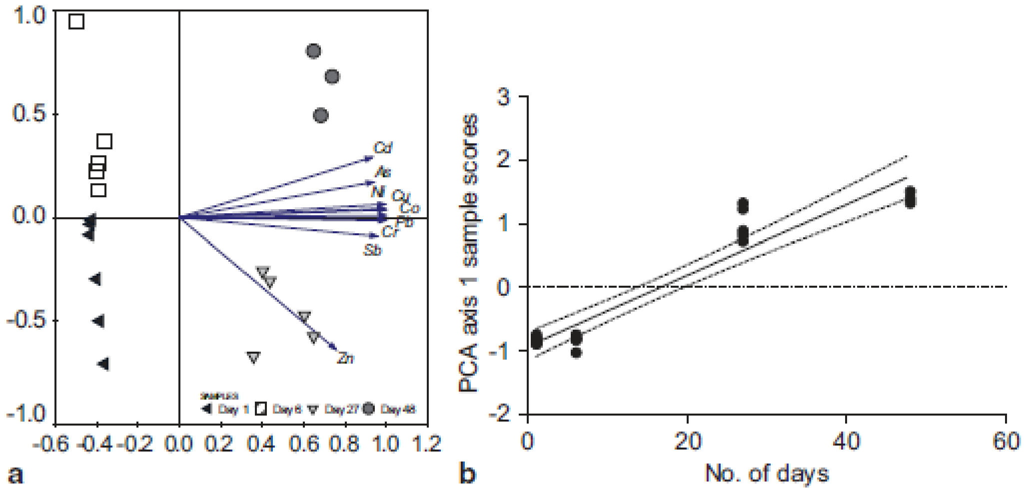


Figure 3.9 a Ordination diagram based on a PCA depicting the time dependent uptake of trace elements in frog eggs and tadpoles. b One-way ANOVA with post-test for linear trend between the days using the PCA sample scores ($r^2 = 0.8$, $p < 0.0001$)

Grung et al. (2016) analysed tissue samples of tadpoles collected from 3 sedimentation ponds for the 16 EPA PAHs. The PAH levels in tadpoles were higher (1–3 ng g^{-1}) than that observed in dragonflies/damselflies (0.2–0.4 ng g^{-1}) but considerably lower than those observed in plants (see Section 3.1.2.5). This indicates either a lower bioaccumulation potential in tadpoles or possibly a biotransformation mechanism for the PAHs.

Jumeau (2017) has quantified the specific abundances and species richness of a range of amphibians in 82 stormwater ponds located in the Bas-Rhin region of France where the road density is 1.9 km/km^2 and consists of 240 km of primary roads and 3654 km of secondary roads. Amphibians, including rare and protected species, were observed in 84% of the stormwater ponds compared to 93% of the semi-natural ponds, which were at least 1 km from a road. However, the occurrences across all species did not differ significantly between the two types of ponds and two species, the European green toad (*Bufo viridi*) and the Marsh frog (*Pelophylax ridibundus*), were more predominant in the stormwater ponds.

3.2 Groundwater impacts

3.2.1 Introduction

Groundwater is a major source of public water supply for many urbanised areas of Europe, and it’s quality status is therefore of considerable significance to all EU Member States. For example, in the urbanised southeast region of England more than 70% of the public supply comes from groundwater sources. Groundwater directly supplies about 25% of London’s water and more indirectly meeting about 60% of minimum surface river flows. Similar situations are found in other parts of the EU with, for example, Portugal having 62 major aquifer systems supplying nearly 5000 m^3/capita/day which represents some 63% of the total national drinking water supply (EASAC, 2010). It is concomitant that potentially deleterious sources and pressures which might intervene to prejudice good groundwater quality be highlighted in order to establish appropriate planning and management control measures.

It is widely recognised that surface sealing (i.e. impermeable surfacing) represents a substantial and major anthropogenic intervention to the hydrological cycle. Such human intervention occurring from urbanisation and transportation drainage infrastructure provision can result in the amendment and disruption of natural surface water flow systems as well as leading to increased flow volumes and associated elevated pollutant concentrations and mass loads to receiving waterbodies (Barrett et al., 1993).

Sustainable Drainage Systems (SUDS) have been developed and implemented across European urban areas to intercept such excess surface water flows and pollutants with infiltration-type facilities being now common drainage design controls for highways and motorways to reduce surface water flood risk and to minimise receiving waterbody pollution (Berger and Ruperd, 1994; Highways Agency, 2009). Infiltration trenches/basins, soakaways, swale channels and grass buffers are all designed to receive stormwater flows from impermeable highway surfaces and recharge local groundwaters by slow infiltration. It is therefore possible that urban/highway groundwater aquifers might become vulnerable to deterioration in water quality if SUDS control schemes are not designed and/or maintained properly (Ellis, 1997).

A risk assessment characterisation undertaken as part of the EU Water Framework Directive (WFD) suggested that up to 25% of groundwater in England and 7% in Wales were at risk of failing environmental objectives and good chemical status by 2021 (Environment Agency, 2014). Diffuse pollution from urban and transport source pressures are deemed to be collectively responsible for some 13% of this national aquifer vulnerability (Environment Agency 2018). France and Spain similarly report 39% and 35% respectively of their groundwater sources to be at risk primarily from over-exploitation and anthropogenic pressures which include urbanisation and associated transport-related land use activities (EASAC, 2010). There are three basic ways in which such transport-derived risks can arise and by which highway runoff can enter and contaminate groundwater:

(i) by sedimentation and filtration to the vegetation and ground surface and
(ii) follow-on surface infiltration and percolation into the soil
(iii) direct sub-surface “injection” into the unsaturated zone and underlying groundwater zone

3.2.2 Highway soil contamination

There is a considerable and longstanding literature covering the theme of highway soil contamination resulting from a combination of direct carriageway wash-off and deflationary/aerial pollutant deposition onto adjacent roadside soils. Early work undertaken in London, UK by Little and Whiffen (1987) estimated that 22% of total vehicular lead emissions were deposited within 10m of the roadside declining to ambient background levels beyond 20 - 30 m. Numerous other studies (Milberg et al., 1980; Warren and Birch, 1987; Lagerwerff and Specht, 1970; Ward, 1990) have all reinforced this conclusion. Even very thin surface soil layers (20 – 25 mm) are capable of capturing and retaining highway runoff contaminants (Waller et al., 1984) and preventing any further downward migration. Lysimeter studies of highway runoff pollution at four sites across the US suggested that sodium and chloride were the only pollutants that might deviate from the observed inverse relationship between pollutant concentration and depth/distance (Kobriger and Geinopolos, 1984).

Site sampling typically exhibits an exponentially decreasing trend of pollutants with increasing distance from the road edge (Aljazzar and Kocher, 2016). Figure 3.10 illustrates the spatial distribution of heavy metals in the upper (0 – 10 cm) soil horizons with increasing distance from the road edge recorded for heavily trafficked (>70,000 AADT) German motorways. Pollutant deposition rates were found to be higher on the leeward side of the motorways essentially due to deflatory dust and spray from the carriageways. Beyond 2 – 3 m from the roadside edge, pollutant concentrations decline to a near constant (but elevated) level to fade into ambient background levels by 20 – 30 m. Typically pollutants such as metals and NaCl at

1 m distance are 4 – 5 times higher than concentrations found at 10 m distance. Within the inner 2 m roadside strip, soil metal levels are substantially higher than most national "precautionary" levels for surface soils. A review of 37 case studies undertaken by 7 EU Member States all showed heavy metals declining exponentially with distance from the highway edge with concentrations in the initial 0.5 m depth approaching "intervention" or "precautionary" levels (Leitao, 2005). Soil pollutant concentrations adjacent to 14 highway sites in Ireland did not reach Dutch "intervention" or "target" levels with the exception of lead levels from remnant leaded fuel deposition (Bruen et al., 2006). De-icing applications however, did generate elevated chloride concentrations exceeding soil "intervention" levels (up to 5 times higher), which further facilitated the migration of adsorbed metals.

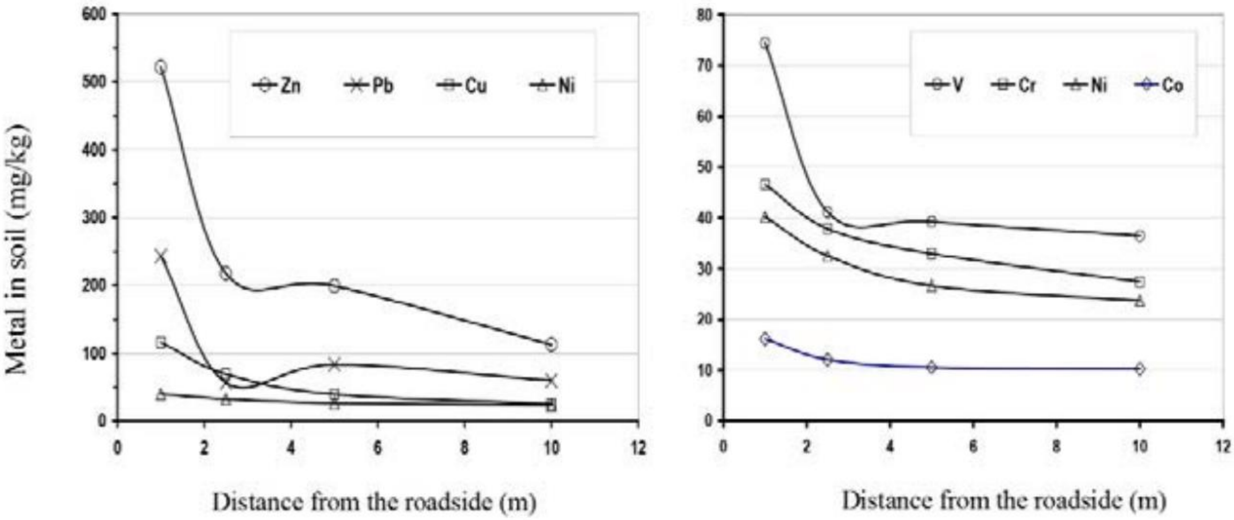


Figure 3.10 Spatial distribution of heavy metals in surface soils with distance from road edge of a German (A61) motorway (after Aljazzar and Kocher, 2016)

The majority of field studies also suggest that soil pollutant concentrations decline with depth as illustrated by the profiles shown in Figure 3.11 for State Route 25 highway at Plymouth in Massachusetts, US (Rotaru et al., 2011). Soil bacterial diversity follows this depth decrease pattern although highway de-icing agent applications would seem to be the predominant control on the subsurface microbial community. Very similar results have been reported for soil and road dust toxicity tests conducted on highway runoff in Melbourne, Australia (Pettigrove et al., 2007) and Japan (Watanabe et al., 2011), even when macroinvertebrate species reached LC50 mortalities within 24 hours following exposure to filtered soil-liquid samples. The principal responsible toxic pollutants were shown to be TPH, metals (especially Zn and Cu) and NaCl. A review of highway pollution in seven EU Member States, whilst finding little evidence for groundwater contamination, did report that elevated metal concentrations occurred in surface soil horizons adjacent to the roadside (TRL, 2002). Nevertheless, little further evidence was forthcoming to substantiate any downward leaching potential into the lower soil horizons or into the unsaturated zone. The only exception was the downward migration of NaCl following application of de-icing salts as well as enhanced metal concentrations as the chloride facilitated the movement of adsorbed metal species.

Studies on 5 major German highways found high pollutant concentrations in the upper 5 cm of soil within a distance of 2 m from the highway edge but which rapidly decreased with depth (Dierkes and Geiger 1999). At 10 m distance and below 0.5 m depths, pollutant concentrations did not exceed German or European drinking water standards. Cadmium exhibited the most significant downward mobility but there was no evidence for any breakthrough. Backstrom et al (2004) noted that there was a marked tendency for aqueous phase metals (such as Cd and Zn) to increase in the roadside soil profile as a response to ion exchange which was enhanced by the formation of chloride complexes.

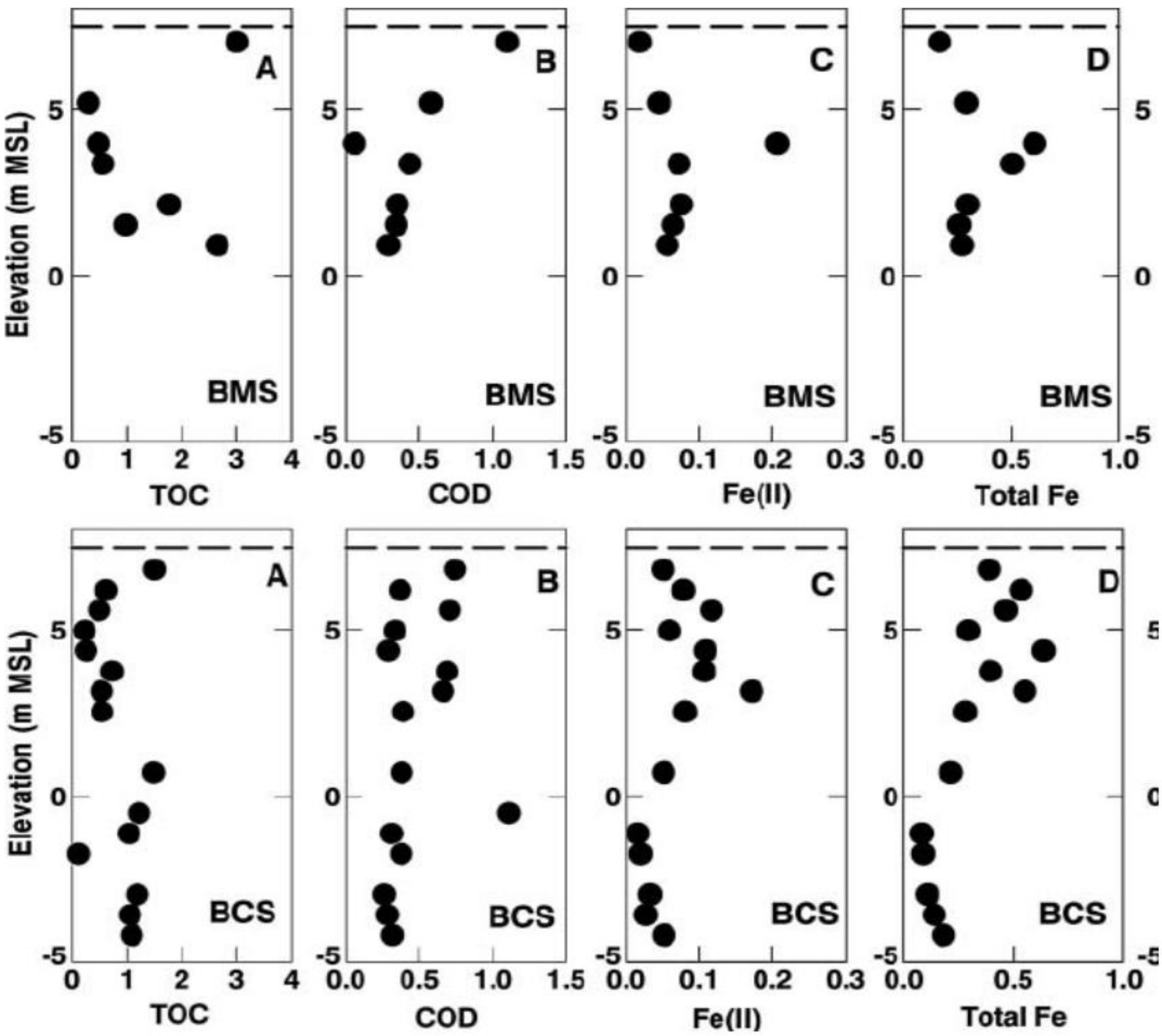


Figure 3.11 Vertical soil profiles showing distribution of highway pollutants with depth for two highway sites (SR25) at Plymouth, Massachusetts, US (after Rotaru et al., 2011).

Toxic leaching tests undertaken by Boothroyd (2008) on soil samples taken from motorway (M1) sites near Nottingham, UK also showed rapid decline of pollutants with depth and only very slow progression of pollutant fronts with depth over time. For sandy soils having a low sorption capacity and high permeability, there was some tendency towards marginal exceedance of relevant drinking water quality maxima and 95 percentile standards with increasing depth. However this was still restricted to the upper 1m of the soil profile. Based on this scenario it was concluded that direct infiltration of polluted highway overland flow is much more likely to play a more significant role in groundwater contamination than soil leaching (Dawson et al., 2009). The investigations of Mangani et al., (2005) on overland flush stormwaters draining the SS73bis highway (18,441 AADT) near Urbino in Northern Italy concluded that such “over-the-shoulder” flows to adjacent soils masked specific fingerprints of pollutant sources in the soil horizons which represented a complex mix of line and point sources (atmospheric aerosols, building, cement, power and road runoff).

3.2.3 Highway drainage sediment contamination

Highway drainage controls including infiltration trenches/basins, soakaways, swales, filter strips and other bio-infiltration systems have the capacity to capture and retain contaminants in basal sediments or onto basal infiltration media (Pitt et al., 1999). When substrate levels reach saturation concentrations in the drainage control device, there is potential for a time-release breakthrough and downward migration of pollutants into the unsaturated and groundwater zones. Additionally, the existence of organically enriched sediment (10% - 20%) exerts a strong affinity for non-ionic compounds which will bind hydrocarbons. Figure 3.12 illustrates vertical profiles beneath a highway infiltration basin draining a motorway carrying 37,000 AADT in Switzerland which confirms the retention of elevated PAH and Cd concentrations at depths of 0.5 m in association with elevated TOC and DOC (Mikkelson et al., 1997); total metal concentrations were found to exceed soil quality standards. It was suggested however, that soluble species including de-icing salts, pesticides, Zn etc., were able to pass directly through the infiltration system.

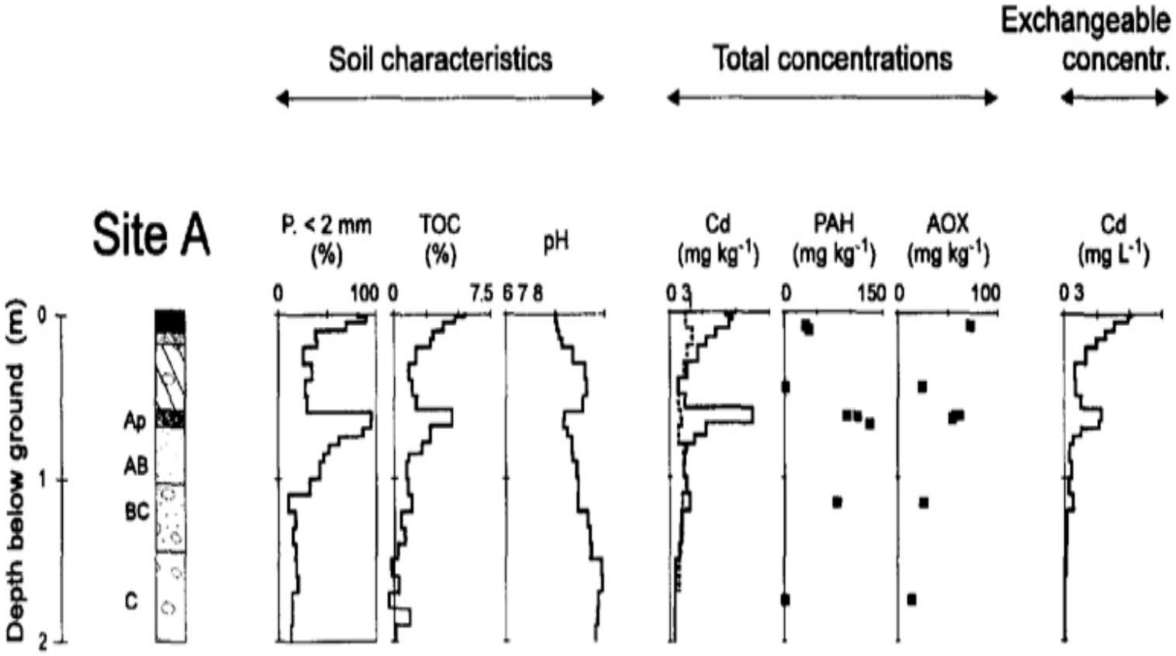


Figure 3.12 Vertical profile beneath a highway infiltration basin in Switzerland (after Mikkelson et al., 1997)

A 30 year old infiltration basin in Lyon, France showed little mobilisation or downward transfer of pollutants from accumulated basal contaminated sediment which clogged the bed of the drainage basin (Datry et al., 2004). Metals and hydrocarbons remained adsorbed to the basal organic particulate. However, during intense rainfall events there was evidence of mineralisation of the organic sediment which led to elevated phosphate and DOC concentrations (4 – 5 times higher than the control) down to depths of 2 – 5m. Leaching and/or enzymatic hydrolysis of the organic sediment following extended contact times, was thought to be the cause of the event-based downward migration. Very similar data and conclusions were reported by Winiarski et al., (2006) in their studies of the same infiltration basin in Lyon. However they stressed that the drying out and "cracking" of basal sediment during extended dry weather periods led to the development of preferential flow paths which enabled a "pumping" of contaminants to greater depths during subsequent storm events. Investigations on basal sediments of infiltration devices draining four major highway sites in southern England found elevated Zn, Cu and Pb which were 6x, 20x and 3.5x greater than adjacent uncontaminated rural soils as well as PAH (4x), especially pyrene and fluoranthene (HCG/HRG, 2009a). The additional occurrence of antimony and barium pointed to the accumulation of traffic-derived pollutants over a 20 – 25 year period.

Seeded incubation testing of the basal sludges (averaging 424 – 6011 mg/kg TPH) of a 10 year old Scottish highway SUDS installed on the M74 south of Glasgow (infiltration basin), the A8000 at Edinburgh (grass swale) and the A90 near Edinburgh (grass buffer strip) confirmed rapid degradation rates for low molecular weight (LMW) compounds (Jefferies and Napier, 2008). The incubation tests suggested in-situ degradation of up to 80% of the initial TPH levels over a period of 224 days, although there was a persistence of high molecular weight (HMW) species.

A wider review (El-Mufleh et al., 2014) of French infiltration basins in Nantes and Lyons which have operated over long time periods (>10 years) confirmed the findings of the earlier French work (Winiarski et al., 2006) and emphasised the importance of regular maintenance practice (and sediment removal) to prevent potential contamination of shallow groundwater sources. Basal clogging has long been regarded as a major issue for the operational efficiency and long term performance of French infiltration basins (Raimbault et al., 1999) and up to 50% of such highway drainage control facilities were reported as being non-functional. A similar proportion of Scottish infiltration facilities were found to be unsatisfactory with 25% of this total rated as having failed (Schluter and Jefferies, 2005), with a further 30% reported as being partially blocked. The clogging mechanism appears to develop at the crushed stone media/soil interface at the base of the infiltration facility where high density fine particulate acts as a "plug" to seal the pores and interstices (Siriwardene et al., 2005). It was estimated by Bergman et al (2011) that two 15 year old infiltration trenches in Copenhagen, Denmark would lose up to 60% loss of their volumetric capacity over a projected 100 year lifetime. Analysis of infiltration basin sediments of four major highways in southern England indicated that whilst being classified non-hazardous as "gully" or "oil separator" waste, criteria in respect of landfill waste would be exceeded for TOC and hydrocarbons (HCG/HRG, 2008). Such waste would require further pre-treatment prior to any disposal acceptance. The report recommended further investigation to identify pollutant accumulation and degradation over time and that systematic and regular maintenance procedures should be implemented for all UK highway infiltration systems. For example, under the German Water Association (ATV 2002) technical standards, it is required that highways carrying more than 15,000 AADT should pre-treat highway runoff prior to surface infiltration and that annual monitoring and inspection be carried out.

Very similar outcomes to those for infiltration basins and filter trenches have been found for highway swale channels which suggests that there is little evidence of any undesirable pollutant constituents reaching the underlying water table (Howie and Waller, 1986) even when variable concentrations occur in the unsaturated zone beneath the swale. A modelling approach to assessing the vulnerability of groundwater to seepage drainage from contaminated sediment accumulating in highway swale drainage systems by Revitt et al., (2017), indicated that the adsorption of more soluble pollutants on basal particulate could increase the potential for downward migration especially of metal species such as Zn, Cd and Cu as well as for PAH. The availability of a basal filter layer to enhance attenuation and degradation (Hatt et al., 2007), as well as regular monitoring and maintenance, was recommended to provide a long term safeguard for groundwater quality. It is clear that infiltration control facilities need to be regularly monitored using a combination of mass balances, in-situ measurements of soil water concentrations and pollutant degradation rates if they are to remain effective highway drainage devices.

In terms of groundwater protection, the maintenance of highway infiltration control devices is essential as a regular pro-active operation with the primary objectives being not only ensuring design volumetric capacity and hydraulic performance but also concern for long term water quality and seepage to groundwater. There is no obvious effective means of extracting or recovering pollutants from the basal filter sludge and once blockage occurs, the entire system has to be replaced. Such replacement on major highway filter drains can be as frequent as every 2 – 3 years (Ellis and Rowlands, 2007). It is quite possible for adsorbable pollutants to accumulate in basal sediment of infiltration systems and eventually reach critical environmental concentration levels. However, whilst such contaminated basal sediment might serve as a sorbent "reservoir", it is much more likely that it presents more of a solid waste disposal problem than a significant groundwater risk (Ellis and Rowlands, 2007).

3.2.4 Highway Drainage Contamination of Sub-surface Groundwater

3.2.4.1 Introduction

It has long been acknowledged that the unsaturated (vadose) zone has a significant role in transporting, attenuating and mediating pollutants which migrate down from surface and sub-surface highway drainage systems (Pitt et al., 1994). In this respect it serves to protect underlying groundwater systems by preventing and limiting the downward transmission of source pollutants (Cassiani et al., 2006). This role is recognised in the EU Groundwater Directive (2006) and the related Daughter Directive (GWDD) and is implemented through national Member State regulations. Unfortunately, only rather limited literature exists which examines the transport and fate of highway pollutants within the unsaturated and saturated groundwater zones with the possible exception of de-icing salts. Infiltration devices are designed to provide direct groundwater recharge and downward basal seepage will necessarily interact biochemically with the unsaturated zone as subsurface flow is drawn down by gravity and pressure forces.

Kayhanian et al (2007) have asserted that the groundwater impact is largely a function of the total storm event rainfall and resultant pollutant mass loading to the infiltration basin which in turn drives downward diffusion to and through the unsaturated zone i.e. net infiltration rate applied per unit area. HCG/HRG (2009b) in their review of UK highway groundwater contamination suggest that it is also important to consider pathway risk assessment as a function of the differing lithological layers (which control preferential flow) within the unsaturated zone in conjunction with biodegradation rates and immobilisation/decay factors.

Figure 3.13 summarises the principal pollutants and transmission processes contained in the conventional groundwater source-pathway-receptor modelling approach to highway runoff which illustrates the complexity of the dynamic interactions which need to be addressed to provide an accurate and robust prediction of groundwater quality both in space and time. It is almost inevitable that there will be considerable site and event-based specificity associated with groundwater impacts with a variable degree of uncertainty as to the real time outcomes.

3.2.4.2 Dual-porosity flows

One major study of highway runoff contamination of the local unsaturated zone is that reported by Robinson et al., (1999) of large lagoon-type soakaways on the M25 London Orbital Motorway (116,900 – 133,905 AADT) at Junction 20 (Kings Langley, Bucks) sited over the Chalk aquifer. Both drainage waters and accumulated bed sediment recorded elevated pollutant levels (Na, Cl, HMs, HCs and PAHs) exceeding “intervention” standards and WHO drinking water guidelines up to 50 times. Despite these high levels, all parameters in the adjacent abstraction wells up to 3 km away from the site were within drinking water limits. Robinson et al (1999) concluded that the low recorded concentrations at the receptor wells could be accounted for by the double-porosity nature of the underlying fractured chalk with storm event discharges being dispersed and diffused into and retarded by the immobile chalk. In addition the aquifer abstraction zone afforded a large dilution capacity of the order of 10^7 – 10^9. However it was also considered that the continuous accumulation of pollutant concentrations in the chalk matrix had the long term potential for deep groundwater impact particularly for those pollutants such as PAHs and soluble metals which possessed low allowable limits.

Diffusion flushing from fracture conduits into the matrix occurs within both the unsaturated and saturated zones which means that the bulk chalk matrix serves as a long term pollutant “reservoir”. Thus there is always the possibility of chlorinated hydrocarbon breakthrough that might be initiated during intense storm event flushing and which highlights the potential of the aquifer to be vulnerable to highway spillage incidents. Given the fractured/fissured nature of chalk and karst lithologies, recharge runoff transit time can be very rapid. Tracer tests undertaken on soakaway drainage at the M1/M25 Junction of the Outer London Orbital motorway in Hertfordshire, UK indicated potential travel times of 1 – 2 km/day via fissured flow through the unsaturated zone (Price et al., 1992). The maximum recorded tracer speed of conduit water in the Chalk was 100 m/hour; tracer travel velocities varied between 100 m/d to 3 km/day although there was a high dilution factor of $10^7 – 10^9$. On the basis of the tracer behaviour, it was estimated that pollutant concentrations reaching abstraction wells some 3 km away from the soakaway injection point were likely to be about 4 µg/l for every tonne of pollutant arriving at the soakaway structure.

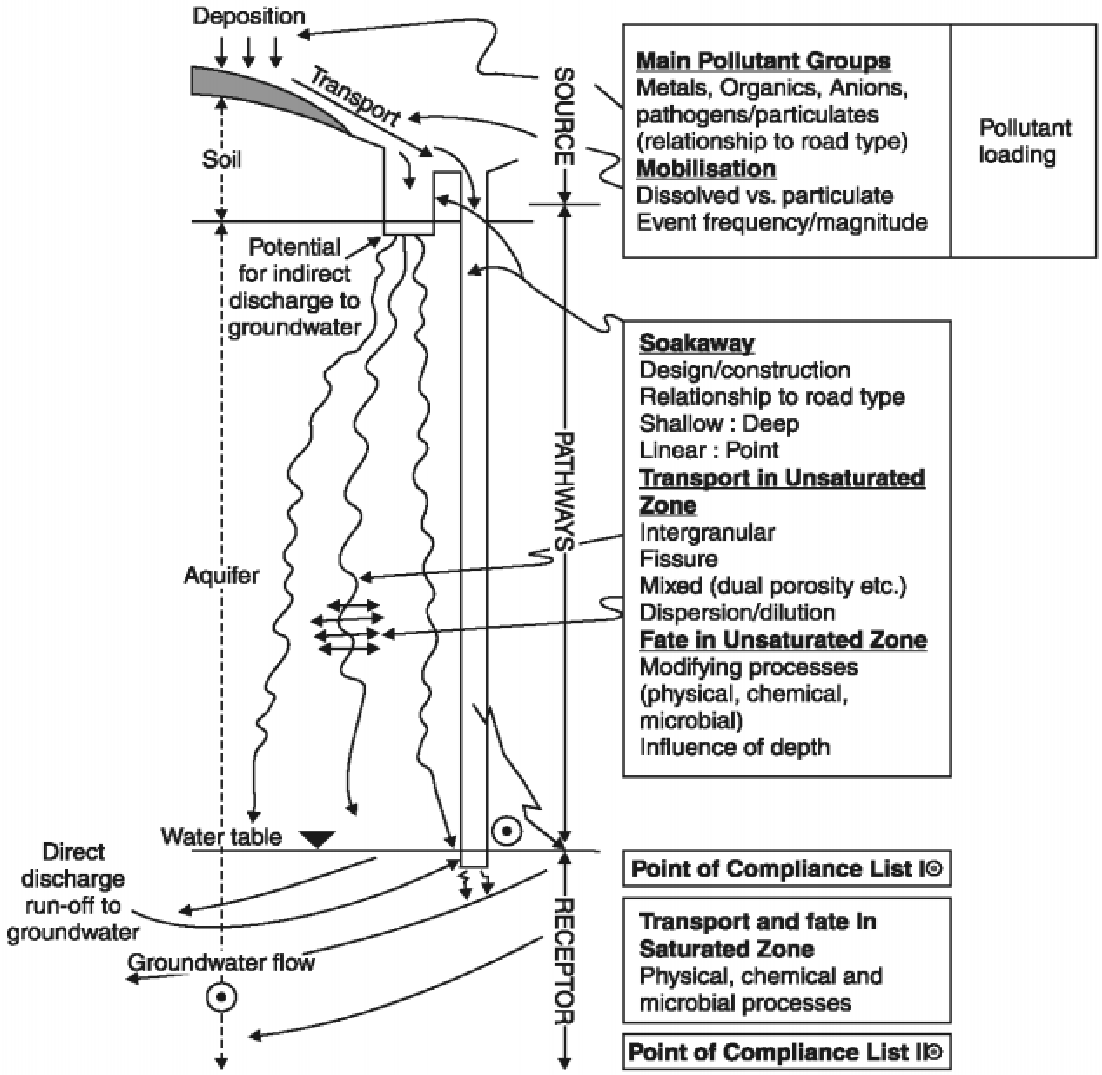


Figure 3.13 Groundwater source-pathway-receptor model of highway runoff (after Highways Agency, 2009)

Once the infiltration rate exceeds the hydraulic conductivity of the infiltrating matrix, flow will move directly down the chalk fissures. This pattern has been confirmed by studies of highway soakaways in Germany (Zimmerman et al., 2004) where release and transmission of large volumes of discrete, dense non-aqueous phase liquids (DNAPLs) was noted into underlying strata. In such aquifer systems the storage capacity for advective downward water is provided by the rock matrix whilst fractures provide the dominant regional flow path. Diffusion moves water into and out of the matrix (Figure 3.13). This diffusion process has an important effect on the form of the breakthrough curve at the receptor point.

The input parameters for dual-porosity modelling are therefore the matrix porosity, the effective diffusion coefficient as well as fissure width, separation and velocity. Double-porosity modelling suggests fissure velocity to be the most sensitive parameter with output concentrations typically exhibiting multiple pulsed peaks over time following a rainfall (or spillage) event. However, given the very high matrix porosity of the chalk and similar karst lithologies, the dilution effect of the diffusion in the dual-porosity system between the soakaway injection point and the down-gradient abstraction well is relatively small. This reduces the risk of down-gradient well contamination and a number of studies have concluded that highway runoff constituents may not present significant sources of pollution to karst aquifers (Donaldson, 2004) due to this large dilution capacity. Nevertheless, this does not negate the higher risk posed by highway spillages as the potentially high transmission rates of the fractured rock can render the receptor point highly vulnerable. In addition, the large matrix storage can serve as a secondary "reservoir" source of chronic contamination at a later date. Although such dual-porosity processes may be operating, the high speed of fracture flow may well prevent significant exchange within the unsaturated zone between fracture and matrix water. It is also the case that given the semi-karstic properties of the chalk, the often used equivalent porous media (EPM) assumptions for groundwater modelling may have little validity.

3.2.4.3 De-icing salts

There is a large literature concerning the impact of de-icing salts on groundwater which all emphasise the potential for significant contamination of aquifer quality. Studies of the SR25 highway (28,000 AADT) near Plymouth, Massachusetts, US observed high down-gradient concentrations of calcium magnesium acetate (CMA) and road salts to be an order of magnitude greater than up-gradient concentrations within a 15m thick unsaturated zone (Granato et al., 1995). This increase occurred in association with a significant decrease in down-gradient pH values. Strong cation exchange together with acidification and mineralisation collectively resulted in Cl, Cd and Cu levels exceeding both secondary and primary drinking water standards. Williams et al. (2000) recorded chloride contamination levels in 23 groundwater springs in the Greater Toronto region of southern Ontario, Canada resulting from winter application of road de-icing salt ranging between 2 – 1200 mg/l. Clear seasonal peaks in chloride concentration were related to periods of highway de-icing salting and toxicity testing showed that sensitive amphipod species such as *Gammarus* died within 24–48 hours following exposure to such spring waters. Howard and Haynes (1993) reported that metropolitan Toronto highways annually received more than 70,000 tonnes of NaCl road de-icing chemicals of which as much as 50% – 60% of the applied chloride enters temporary storage within the shallow unsaturated zone. It was asserted that such application loadings might result in steady-state groundwater concentrations reaching 426 (±50) mg/l within a 20 year period, which would be double the drinking water standard. Maximum recorded groundwater chloride concentrations reached 14,000 mg/l which would be 50 times the standard. Mass balance analysis by Perrera et al., (2013) indicated that as much as 40% of highway salt applications in the Greater Toronto region entered shallow groundwater via fast fissured flow pathways during the winter period.

The investigations of Kunze and Sroka (2004) of the groundwater impact of winter runoff in Ohio, US for highways having between 15,000 – 20,000 AADT subject to annual salt (NaCl) applications of between 4,599 – 10,766 tonnes showed that only 3 of the 8 highway sites to be affected by the application of de-icing chemicals. Only relatively minor traces of dissolved chloride (means 24 – 43 mg/l) above a background concentration (means 13 – 23 mg/l) were determined in the shallow (< 15m) unsaturated groundwaters. Average groundwater velocities were 0.05 m/day with a median aquifer bulk hydraulic conductivity of 1.65 m/day but there was considerable local variability in these parameters which effected the final sampled groundwater concentrations. Infiltration control basins alter ambient advective processes and patterns by displacing hydraulic pathways downward and augmenting vertical mixing by imposing a periodic vertical fluctuation to the ambient flow fields as reflected in Figure 3.14. In general, the Ohio groundwater responded to rainfall events within one day (Kunze and Sroka, 2004) indicating quite rapid transmission within the unsaturated zone with only mild attenuation down-gradient as indicated by Figure 3.14. Mean concentrations at the down-gradient well (AS47) were about 3 times the up-gradient or background concentration at this site.

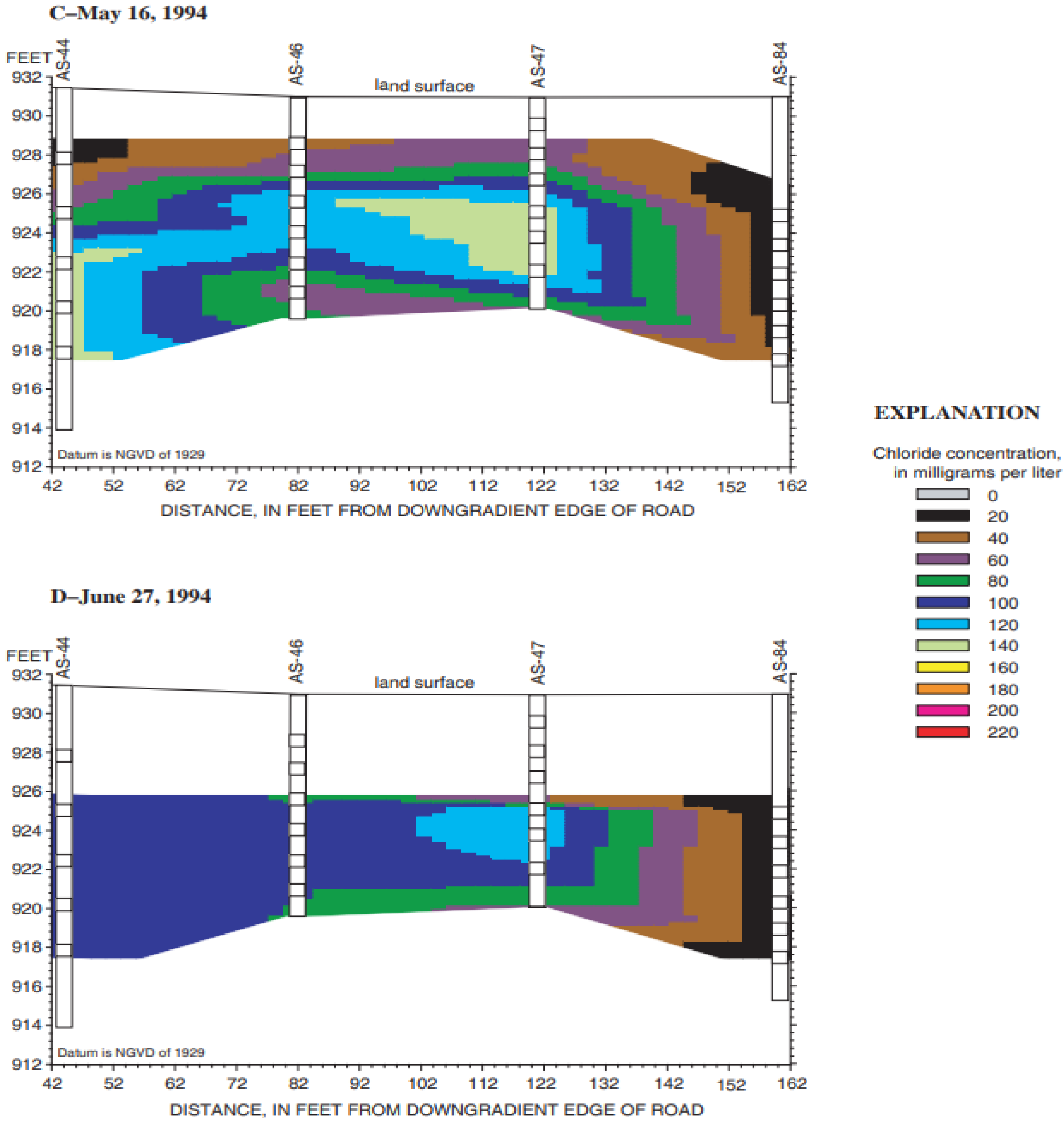


Figure 3.14 Iso-plot Profiles of Chloride Concentration for Highway SR3, Ashland, Ohio, US (after Kunze and Sroka, 2004)

It has long been recognised that temporal variation between surface water levels and abstraction well levels can be very short, sometimes within hours of each other, indicating that in karst-like strata, surface and groundwaters are intimately interactive and that fast fissure flow is dominant in the groundwater system (Van Brahana., 1998). Figure 3.15 shows a cross-section illustrating the maximum development of a lateral and downward plume-like chloride dispersion from a highway injection source which is typical of the pattern associated with highway contamination of groundwater and which is generally confined to relatively shallow depths (<10 – 20m) and extending some 100 – 300m from the line-source. The decrease in chloride with increasing distance along the groundwater flowpath supports the conclusion that mechanical dispersion (i.e. mixing) substantially affects the distribution of the de-icer related compounds in the aquifer. Quite frequently sustained monitoring data will confirm that chloride concentrations at the down-gradient water table level will not return to background conditions, implying a gradual accumulation over time to steady-state (Watson et al., 2002).

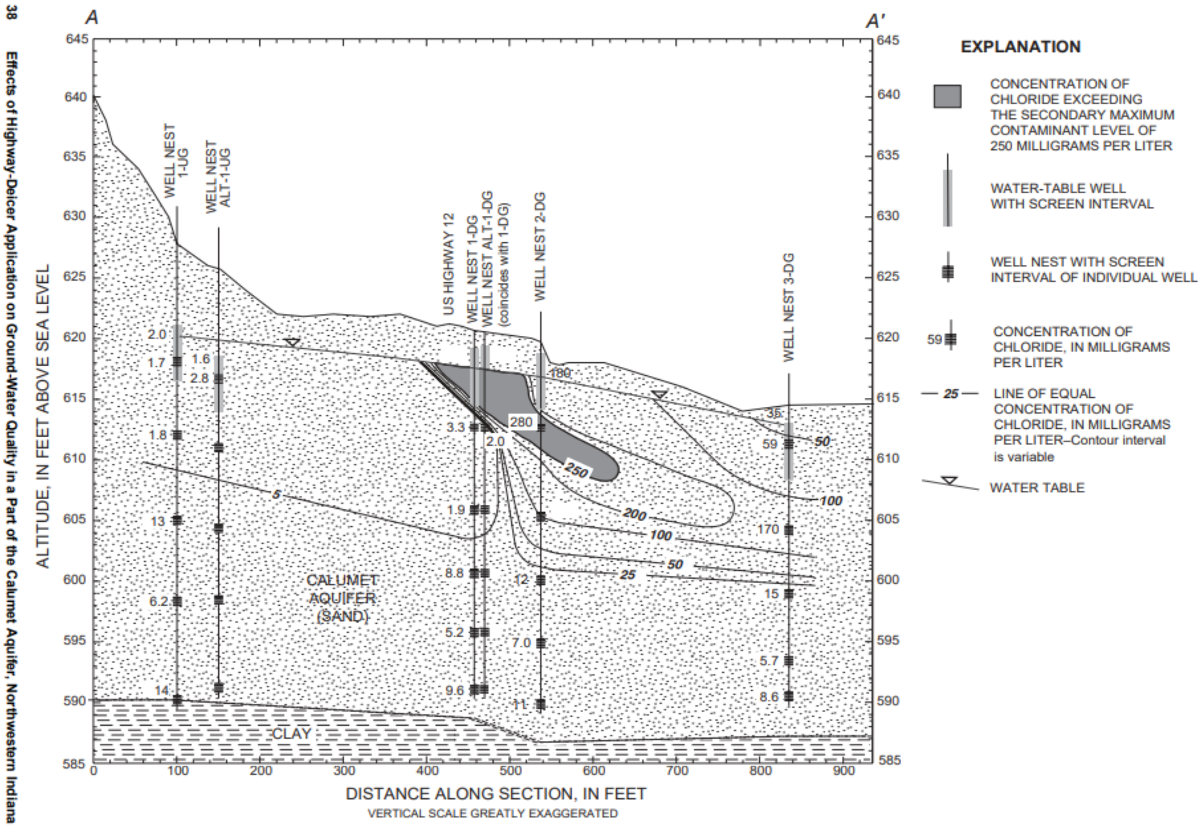


Figure 3.15 Groundwater chlorine concentrations beneath a highway (after Watson et al., 2002)

The HCG/HRG (2009b) review of UK highway contamination conceded that elevated winter groundwater chloride levels resulted from de-icing operations and induced concurrent mobilisation and release of bound or sorbed organic pollutants and more soluble metal species (Cd and Zn) retained in infiltration sediments and matrix pore-waters. The downward pollutant migration typically follows a colloidal plume-like dispersion pattern through the unsaturated zone down to depths of 5 – 8m (see Figure 3.15). Investigations of de-icing salts in runoff from the M5/M42 motorway interchange southwest of Birmingham, UK confirmed that chloride washoff from the highway to the receiving stream leached down to the underlying aquifer (Rivett et al., 2016) in such as plume-like form. The leakage source (equivalent to 63 tonnes of applied salt) was estimated to account for between 21% – 54% of the 70 tonne supply well annual increase (over baseline); maximum chloride concentrations of 13,500 mg/l were recorded in the groundwater. The mass flux chloride shortfall was inferred as being the result of retention by adsorption at or close to the stream and basal sediment infiltration location. However, the pan-European highway POLMIT project showed that whilst chloride levels frequently exceeded national soil and groundwater intervention levels (by more than 5 times), the high winter load is rapidly reduced by dilution in the unsaturated zone to below threshold levels during summer and autumn periods (TRL, 2002). PAHs and HMs (especially Cd, Zn and Cr) were also found to marginally exceed intervention levels in association with the seasonal increase in chloride. During downward contaminant migration only dilution moderates the concentration of the chloride ion. Sodium ion concentrations by contrast are often significantly modified by ion exchange which releases into solution divalent cations, notably calcium.

3.2.5 Process and Data Limitations

The concept that infiltration provides a "treatment" process is simplistic in that any mass removed from solution must either remain stored within the soil compartment or be leached out i.e. it really serves as a mass storage technology. Exchange capacity and sorption processes do not remain fixed over time or space, being highly dependent on local prevailing soil and solution conditions. Any changes in the water quality character infiltrating a site can potentially change the geochemical conditions, leading to the possible release of the sorbed mass in the subsoil/sludge.

The conventional design guidance for infiltration systems does not currently sufficiently consider the issue of "facilitated transport" within the unsaturated zone. Fundamental processes such as metal and hydrocarbon complexation (and subsequent transformation and transport) with dissolved and natural organic matter (DOM/NOM) remain highly speculative as do the unsteady hydraulics of repetitive, pulsed cycling in double-porosity strata. There is considerable geochemical evidence that non-ideal solute breakthrough (with long tailing and sharp initial wave fronts) is a normal consequence of natural porous media. NOM-metal complexation possesses retardation factors anything between 4 – 7 times lower than uncomplexed forms. Metal breakthrough can also lag the DOM breakthrough. This could suggest that the soil substrate "cleanses" the metal-DOM complex as it infiltrates, resulting in an initial breakthrough of metal-cleansed DOM followed by a later breakthrough of toxic metal-DOM complex as the soil becomes increasingly saturated with DOM. Size exclusion may also play some part in the co-transport of the metal with DOM. This metal complexation to soluble DOM can control metal mobility, with partitioning to sorbed DOM playing only a relatively minor role.

If such processes are generally valid, then ultimate metal (and soluble hydrocarbon) breakthrough to the underlying groundwater is almost inevitable. It has been argued that the application of compost and other recycled materials as substrate media in porous paving and trench systems can be effective in removing monolayer soluble/colloidal metal species (Seelsaen *et al*., 2005) but they can also leach out high concentrations (> 4mg/g) of DOC. In addition, existing colloid facilitated transport modelling assumes that the relevant partitioning mechanisms follow linear, equilibrium sorption kinetics. The use of such simple modelling assumptions would be inadequate to describe the NOM-metal complexation. Existing background metal concentrations in the underlying soil layers must be important considerations in infiltration design, since metal displacement will occur as a result of competitive adsorption/exchange, and/or dissolution effects posed by the multi-component system. Sub-soils containing concentrations in excess of 20 μg/g for Cu and Pb, 50 μg/g for Zn and 1 μg/g for Cd should be avoided. In addition, organic carbon (OC) content should exceed 0.5% to improve metal attenuation, and minimum depth to any underlying unconfined aquifer should be at least 3m (Pitt et al., 1999).

Reported data is collated from varying climatic and geographic circumstances and hence have been determined under widely differing sampling and analytical regimes. Some data simply report concentration or load averages and/or ranges whilst others are based on systematic regular (e.g. seasonal or weekly) sampling with only relatively few being based on time series storm event data. Many data sets do not refer to detailed receiving water quality impacts resulting from contaminated highway discharges. Pressures and impacts on groundwater and receiving water bodies from such runoff sources are thus largely dependent on local site and time (or event) circumstances and can vary widely in intensity both temporally and spatially. It is therefore difficult to appropriately classify and compare the literature data sets. Such aggregated data may not reflect overall national groundwater condition or status or reflect the specific level of risk posed to individual aquifer quality. There is undoubtedly a justifiable argument for a harmonised pan-European approach to monitoring and reporting of information on the state of urban/highway groundwaters and the specific urban and transport pressures placed upon them in both acute and chronic terms. Monitoring should focus on shallow groundwater and the unsaturated zone in order to provide an early warning system and to provide the basis for effective groundwater modelling tools. This might be achieved through an extension of the existing EUROWATERNET network but this would need the agreement and support from all EU member states to be cost-efficient and performance effective.

3.3 The impacts database matrix: overview and use

The D2.1 database matrix (which can be accessed on the CEDR PROPER website at: www.proper-cedr.eu/) contains details of sixty studies providing further information (e.g. receiving water geology, road characteristics, receiving water characteristics and impact) on many of the papers reviewed within Sections 3.1 and 3.2 of this report. Data reported relate to a range of environmental endpoints (e.g. fish, amphibians and invertebrates) and/or receiving compartments (exceedance of surface water and groundwater EQS; exceedance of sediment quality guidelines). The matrix can be used to locate studies which specifically refer to an impact of particular interest in receiving waters, and the matrix column headings are used as a framework to give an overview of data entered as follows:

- road characteristics (length and/or width: 47 papers; slope: 26 papers; AADT: 45 papers; type of road material: 35 papers)
- receiving water body type (23 papers)
- water body quality status (5 papers)
- application of de-icing materials (35 references)
- receiving water substrate/rock type (identified in 46 papers)
- infiltration rate (referred to in 10 papers)
- depth to groundwater (reported in 23 papers)

The following sections present the results of a simplified contents analysis, presented as a way to synthesise the matrix inputs to identify:

- the number of papers which refer to an identified aspect e.g. infiltration rate
- the way data on each aspect was presented; for example, quantitatively (e.g. m/d; L/s) and qualitatively (e.g. moderate rate of infiltration)
- keywords to enable a user to locate specific papers (using the ctrl F function in Excel)

Road characteristics

- Forty-seven matrix entries refer to the length and / or width of road responsible for generating the runoff under investigation. The most common descriptor is the identification of the number of lanes, with specific width and length dimensions of road area drained co-reported only rarely. Key words: number of lanes
- Forty-five papers provide information on the annual average daily traffic (AADT), with the percentage of traffic consisting of HGVs also commonly identified. Key word: AADT
- Thirty-five entries identify the type of road material. In 29 entries the road material was identified as asphalt, with four studies reporting the use of porous asphalt/porous materials, one study the use of bitumous tarmac and a final study identified materials as ‘impermeable’ only. Key words: road surface; porous asphalt.
- Twenty-six papers refer to the road slope. On the majority of occasions road slope is identified as ‘a gentle slope’ (11 occasions) or as flat (11 occasions). The slope is reported quantitatively (e.g. 4%) in only four entries. Key word: slope
- Thirty-five of the papers refer to the use of de-icing materials. With one exception, all of the studies referred to the use of sodium chloride as the de-icing agent. Seven studies report the load applied, of which three studies also identify the frequency of application. The only alternative de-icing material identified was calcium magnesium acetate in the USA between 2001 and 2005 (Rotaru *et al.,* 2011) before being replaced with sodium chloride in subsequent years. Key word de-icing material

Receiving water characteristics

- Twenty-three of the papers identify the receiving water type. The most commonly identified receiving waters are ground waters (nine studies) and rivers (seven studies), a surprising result influenced by the inclusion of several case studies from the POLMIT research project which focussed on developing understanding of the impacts of road runoff on groundwater (TRL, 2002). Other water body types include streams, ponds and lakes. Key words: river, stream.
- As the EU Water Framework Directive (WFD) applies to waterbodies with a surface area >0.5 km^2, many of the waterbodies monitored are likely to be too small to be specifically considered within this context. This may explain why EU WFD status was referred to in only four of the studies reported in the database matrix. Of these papers, three studies identified the ecological status upstream and downstream of the road runoff discharge point with two of the three studies reporting no change in status between identified sampling points. Key words: chemical status; ecological status.

Receiving water impacts

- As the aim of data collection was to review papers with a focus on the vulnerability of receiving waters to road runoff discharge, all sixty entries refer to the occurrence or absence of impacts within receiving waters. Between them, the studies reported the assessment of a range of receiving compartments (e.g. surface water, groundwater, sediments) with impacts considered in terms of the exceedance of pertinent receiving water environmental quality standards, groundwater threshold values and/or guideline values. A combination of field and laboratory studies where undertaken to understand the impact of road runoff on a range of environmental endpoints including fish (e.g. minnow, roach and brown trout) and amphibians (e.g. frog eggs and tadpoles); invertebrates (e.g. amphipods and rotifers) and plants (e.g. periphyton and pondweed). The results of these studies are comprehensively discussed in Sections 3.1 and 3.2 (see relevant sections). Key words: invertebrates; *Daphnia*; EQS.

Receiving water geologies

- Forty-six matrix entries refer to receiving water substrate and/or rock type, of which 21 studies refer to groundwater. 17 discrete categories of soil type are listed, with the most common soil types identified as sand (5 studies), clay (5 studies), limestone / karst (8 studies). Other substrate types identified include clay/loam, silt/loam, gravel and weathered granite, together with the use of more generic descriptors e.g. alluvial, moraine. Key search terms: clay, karst, sand, loam.
- Twenty-three papers refer to the depth to groundwater / water table either quantitatively (e.g. 10 m); semi-quantitatively (e.g. >100 m) or qualitatively (high groundwater level). Only two papers reported that groundwater depth varied seasonally. Key words: water table; depth to groundwater
- Only ten entries in the matrix were found to refer to the infiltration rate of receiving water substrates at study sites, using both quantitative (e.g. as m/d; L/s) and/or qualitative terms e.g. high rate of infiltration. Five papers referred to groundwater receiving bodies. Keyword: infiltration rate.

3.4 Relative Importance of Contributions of Traffic-Related Pollutants in Relation to Other Identified Pressures at a Site and Catchment Scales

Surface water runoff from sealed impervious surfaces has long been recognised as potentially a significant contributory source of pollution to receiving waterbodies (Hvitved-Jacobsen and Yenisei, 1991). It has been suggested, for example, that between 35% to 75% of the total receiving water metal budget in urban areas might be derived from highway runoff and that the ambient water chemistry can become acutely dominated by event-based highway discharges (Ellis et al., 1987). However, there is only rather limited data to satisfactorily quantify the relative contribution of highway surfaces to total receiving water pollutant loading and is particularly scarce in terms of quantification of traffic-related pollutants in relation to other line and point sources at either at either site or catchment scales.

The UK national risk assessment submitted as part of the WFD reporting requirements (Environment Agency, 2018) indicated some 13% of aquifer vulnerability in England and Wales could be placed against urban and transport pressures with Scotland reporting 11% of urban receiving surface water vulnerability being due to transport-related pressures (Betson et al., 2006). Figure 3.16 illustrates the relative contribution of highway sources to all diffuse pollution sources as recorded for Scotland in the WFD risk assessment survey and it is clear that contributory pollution from highway discharges comprises only a very small amount of the total pollution load from diffuse sources.

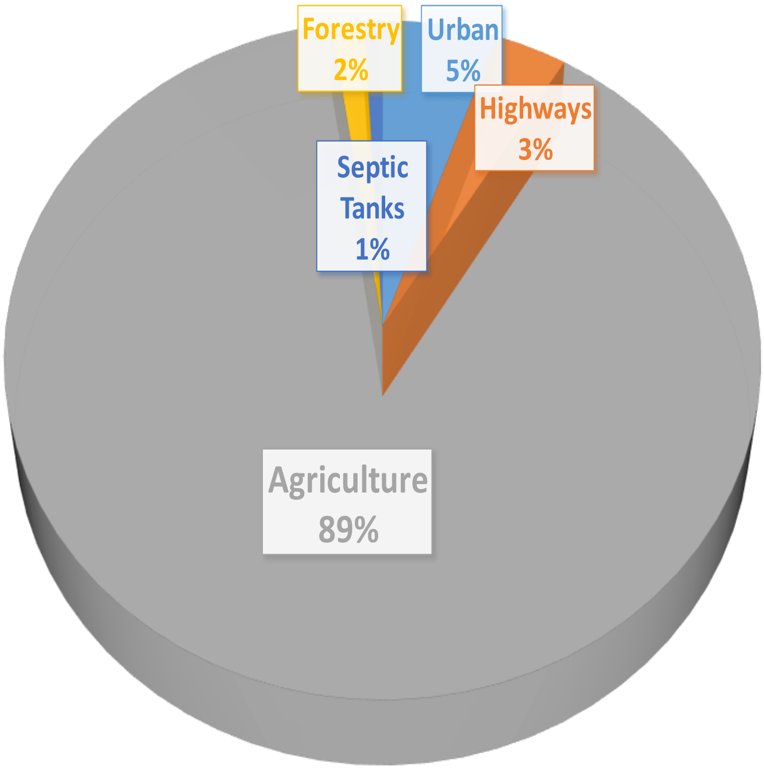


Figure 3.16 Sources of diffuse pollution in Scotland

The same conclusion can be deduced from the data shown in Figure 3.17 where the relative contribution of major highways (including motorways) in the Glasgow area to total highway and urban pollutant loads can be seen to comprise a very small proportion. Major highways contribute only 1.4% of the total Glasgow City urban loadings. When total suspended solids loadings are removed from the data, highways still only contribute 1.7% of the remaining total urban pollutant load.

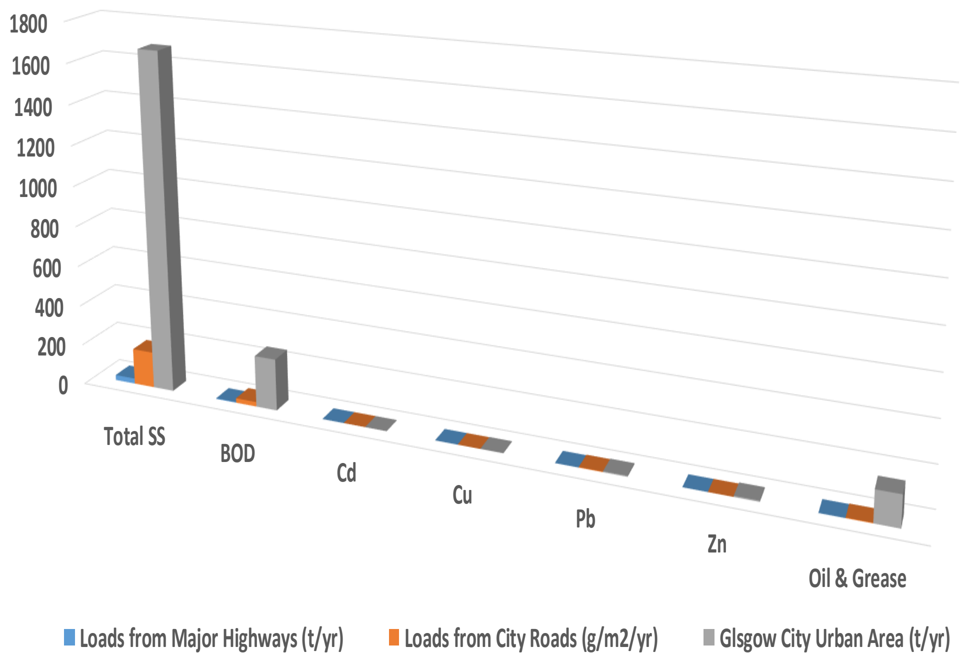


Figure 3.17 Glasgow highway and city pollutant loads

However, detailed mass balance studies undertaken in London did indicate rather higher contributions of highway pollutant loadings than recorded in the Glasgow region (Ellis et al., 1987). Table 3.7 suggests that between 13% - 78% of metals recorded in the receiving stream of a 243 ha residential urban catchment in North London (< 15000 AADT) were caused by highway discharges.

Table 3.7 Metal loading contributions to urban receiving waters in London

Pollutant	Contribution from highways (%)	Estimated loading of particulate associated metals from highway sources (kg)	Soluble-insoluble metal ratios	Estimated annual loading of soluble metal from highway sources (kg)	Estimated annual loading of total metals from highway surfaces (kg)
Cd	46.4	0.07	4.0	0.28	0.35
Cu	78.4	1.35	1.75	2.36	3.71
Pb	47.0	5.4	0.23	1.23	6.63
Zn	13.1	5.08	3.0	15.24	20.32

However, a major problem to be considered when extrapolating such outfall metal source loadings is to account for transformations which occur following removal from the highway surface and which take place in the below-ground gully and pipe system prior to discharge to the receiving stream. Morrison et al., (1988) have shown that large changes in the association (i.e. solid to soluble species) and concentrations of metals can occur between entering and leaving the below-ground drainage system. Thus it is quite likely that the true contribution of highway surfaces to total catchment loading is smaller than suggested in Table 3.7. Other urban mass balance studies such as that of Hoffman et al (1985) for a 0.448 km^2 urban catchment on Rhode Island, US suggested that heavily trafficked (> 100,000 AADT) highways increased metal loading factors by 2715, 11460 and 36 $g/km^2/cm$ rain for Cu, Pb and Cd respectively. These values would yield highway contributions to the catchment outfall loadings of 70.2 kg Cu, 296.5 kg Pb and 0.93 g Cd. Comparison of these values with the data given in Table 3.7 indicates, particularly in the case of Pb, the considerable influence that increased vehicular activity can have on the metal loading distributions.

It is difficult to accurately monitor and measure wet/dry deposition contributions to total highway runoff pollutant loadings. Few investigations use methods that can accurately determine such contributions particularly given the extreme difficulty of “on-highway” processing, resuspension and deflation (Colman et al., 2001). Much better modelling capabilities are needed here to understand and determine atmospheric deposition rates and their contribution to highway runoff loadings. It is highly likely that dry deposition is the more important fraction (Sabin et al., 2005), but particle size is undoubtedly a significant factor in loss/removal rates from the highway surface and this needs further research. Research cited by Huber et al. (2016) reported that presence of highway structures such as noise barriers result in higher runoff concentrations, presumably a function of reduced air circulation leading to the enhanced local deposition of particles and associated pollutants. Comparisons of surface deposition with runoff loads have largely been based on the assumption that “on-highway” transport of deposited materials is conservative but this ignores the effect of induced turbulent resuspension. Determination of ambient deposition and the character and contribution of on-highway pollutant processing remains an outstanding methodological issue.

Studies of chronic contamination of receiving stream sediments in urban areas of Scotland indicate that highway discharges can indeed be a major source of PAHs and other hydrocarbons (Wilson et al., 2005). Four of the nine streams surveyed would have their bed sediments classified as “special” waste under UK regulations. Analysis indicated pyrolytic sources (having low (<10) phenanthrene/anthracene ratios) were ubiquitous together with higher proportions of benzanthrenic combustion sourced substances. However, the source of the latter may well be due to more distant point-based emissions from oil-fired boilers which represent aerial deposition within the highway catchment.

Huber and Helmreich (2016) have attempted long term mass balances of traffic-related emissions and heavy metal runoff loads from highways in Germany. The percentages of directly traffic-related metal emissions (based on calculated total emissions), were 11%, 45%, 98%, 5%,42% and 95% for Cd, Cr, Cu, Ni, Pb and Zn respectively. This would confirm the significance of highway-sourced contributions to total receiving waterbody loadings. The most relevant metals in terms of contamination potential appear to be Cu and Zn, although the persistent chronic accumulation of Zn and other toxic species such as Sb, have major long term implications for the aquatic biota and receiving water ecological status. However, it should be noted that other similar mass balance studies undertaken in Germany by Hillenbrand et al. (2005) recorded even higher traffic-related emission values and source contributions. The difference in outcomes and predicted values are high but perhaps are not surprising given the variability in the monitored data, traffic sources, types and densities as well as varying storm and site characteristics. The highway line source is itself comprised of a complex sub-mixture of traffic derived emissions (exhaust, tyre and brake wear, metal etc.), road furniture/installations (fencing, lighting, barriers etc.) and road materials (asphalt, grit etc.). These sub-sources can lead to a high variability in local pollutant concentrations in runoff discharges which can be further exacerbated by variability in the storm event (intensity, duration, first-flush etc.). Predictive modelling and mass balance becomes difficult and subject to considerable temporal and spatial variability with any statistical relationships being very site-specific.

The specific impact of highway runoff is clearly dependent on the attributes and interactions of the storm event, the highway runoff, the below-ground drainage system and the receiving water. The impact is therefore highly likely to be site and event specific as well as variable in time and space. Most highway pollution studies have established some statistical relationships and possible trends but have failed to establish clear cause-effect relationships between the controlling parameters of the storm event, highway characteristics and traffic densities and receiving water attributes.

4. Vulnerability assessment methods and tools

4.1 Review of tools to predict impacts of road activities on receiving and groundwater bodied near non-urban roads

Outline details of six different modelling approaches for the prediction of highway runoff concentrations are provided in this section. However, only two of the models attempt to address an assessment of the risks posed to the receiving water environment. These are SELDM (see Section 4.1.1) and HAWRAT (see Section 4.1.2). Only the IMPACT model (see Section 4.1.3) adopts a deterministic approach by modelling the processes that influence the movement of pollutants from the construction and repair materials which are typically used in the highway environment. Because of the selectiveness of the assessed materials, this model is fairly limited in its scope but it is the only one of the described models which considers the possible vertical movement of pollutants to groundwaters. Five of the identified models are based on stochastic approaches and have been developed to varying levels of complexity and sophistication. The coupled MT-GA model (see Section 4.1.4) allows the incorporation of a small degree of physical process representation. Two of the modelling approaches (PREQUALE [see Section 4.1.5] and 'Impact of AADT on highway runoff pollutant concentrations' [see Section 4.1.6]) essentially use multiple regression analysis to predict highway runoff pollutant concentrations. All the described models which adopt a data-driven approach are heavily dependent on the authenticity of the data used to develop the model and how relevant this is to the highway environment to be modelled, for example in terms of geographical location, climatic region and consistency with current road usage characteristics.

4.1.1 SELDM: Stochastic Empirical Loading and Dilution Model

The Stochastic Empirical Loading and Dilution Model (SELDM) is designed to transform complex scientific data into meaningful information about the risk of adverse effects of runoff on receiving waters, the potential need for mitigation measures, and the potential effectiveness of such management measures for reducing these risks. The U.S. Geological Survey developed SELDM in cooperation with the Federal Highway Administration to help develop planning-level estimates of event mean concentrations, flows, and loads in stormwater from a site of interest (Granato, 2013). Planning-level estimates are defined as the results of analyses used to evaluate alternative management measures and are recognized to include substantial uncertainties (commonly orders of magnitude). SELDM uses information about a highway site, the associated receiving-water basin, precipitation events, stormflow, water quality, and the performance of mitigation measures to produce a stochastic population of runoff-quality variables. Input statistics for precipitation, pre-storm flow, runoff coefficients, and concentrations of selected water-quality constituents are derived from national datasets. They can be selected on the basis of the latitude, longitude, and physical characteristics of the site of interest and the upstream basin. The user can also derive input statistics for each variable that are specific to a given site of interest or a given area.

Using Monte Carlo methods, SELDM produces the random combinations of input variable values needed to generate the stochastic population of values for each component variable. The dilution of runoff in the receiving waters and the resulting downstream event mean concentrations and annual average water concentrations are calculated. Results are ranked, and plotting positions are calculated, to indicate the level of risk of adverse effects caused by runoff concentrations, flows, and loads on receiving waters by storm and by year. Unlike deterministic hydrologic models, SELDM is not calibrated by changing values of input variables to match a historical record of values. Instead, input values for SELDM are based on site characteristics and representative statistics for each hydrologic variable. Thus, SELDM is an empirical model based on data and statistics rather than theoretical physiochemical equations.

SELDM is a lumped parameter model because the highway site, the upstream basin, and the receiving basin are each represented as a single homogeneous unit. Each of these source areas is represented by average basin properties, and results from SELDM are calculated as point estimates for the site of interest. Use of the lumped parameter approach facilitates rapid specification of model parameters to develop planning-level estimates with available data. The approach allows for parsimony in the required inputs to and outputs from the model and flexibility in the use of the model. For example, SELDM can be used to model runoff from various land covers or land uses by using the highway-site definition as long as representative water quality and impervious-fraction data are available. If a stormwater best management practices (BMP; US term for a sustainable drainage system (SuDS)) is specified, highway-runoff results are produced for both the highway site and the outlet of the BMP.

The Highway Runoff Database (HRDB) is the primary source of highway-runoff statistics and data across the US for use with SELDM and provides the basis for defining runoff quality and quantity at monitored sites and predicting runoff quality and quantity at unmonitored sites. Version 1.0.0 of the HRDB includes data from 2,650 storms for 39,713 EMC measurements of more than 100 water-quality constituents monitored at 103 sites (Granato and Cazenas, 2009). This national water-quality database is continuously being updated and by 2014 included 54,384 EMC measurements of 194 water-quality constituents monitored at 117 sites during 4,186 storm events. It represents a primary source of upstream-water-quality data that is readily available for use with SELDM. The compilation is based on data available in the USGS National Water Information System Web (NWISWeb), a source of water-quality data that can be used to estimate local, regional, or national water-quality parameters (USGS, 2018). As of October 2011, NWISWeb included data for 118,000 stream sites, 2,456 canals, 1,701 ditches, 550 outfalls, 534 wetlands, 211 storm sewers, 145 combined sewers, and 141 pavement sites. However, a review of the data indicates that the number of water-quality samples listed for any given constituent is small for many datasets.

4.1.2 HAWRAT: The Highways Agency Water Risk Assessment Tool

The English National Roads Administration (NRA) and the England and Wales Environmental Agency (EA) in conjunction with the UK Highways Agency have developed the Highways Agency Water Risk Assessment Tool (HAWRAT). The main purpose of this tool is to investigate the effects of road runoff – with and without mitigation measures – on receiving waters. HAWRAT can also predict the statistical distribution of key pollutants in road runoff. The tool considers soluble pollutants, associated with acute pollution impacts, expressed as event mean concentration (EMCs) of dissolved copper and zinc; and particle-bound pollutants, associated with chronic pollution impacts, expressed as Event Mean Sediment Concentrations for total copper, zinc, cadmium , pyrene, fluoranthene, anthracene, phenanthrene and total polycyclic aromatic hydrocarbons (PAHs). The outcomes are aligned with the requirements of the Water Framework Directive risks of pollution to surface waters from highway runoff.

The procedure is included in Volume 4 (environmental assessment) and Volume 11 (design guidance) of the Design Manual for Roads and Bridges (DMRB, 2009) guidance on drainage assessment. In addition to consideration of surface waters, this Design Manual contains assessment methodologies for groundwater discharges and spillage risk, and identifies the circumstances where highway runoff is likely to have a significant environmental impact. Guidance is also provided on the design, construction and maintenance of drainage including Sustainable Drainage Systems (SuDS). DMRB is mandatory for use on all road projects on the strategic road network. Local highway authorities are not required to use the DMRB but do so as it is established best practice.

HAWRAT is a Microsoft Excel application for which a full description and the associated Help Guide and Technical Manual can be downloaded from the Highways Agency Drainage Data Management System (HADDMS). HAWRAT adopts a tiered consequential approach to assessment and can report the results at three different stages depending upon the level of assessment required for any given site. These are:

- Step 1, the runoff quality (prior to any pre-treatment and discharge into a water body) through prediction of the statistical distribution of key pollutant concentrations in untreated and undiluted highway runoff over a long release period. The applied statistical model has been developed based on a ten year rainfall series relevant for the chosen site and its climatic region. The results are reported on a pass/fail basis against established toxicity thresholds.
- Step 2, in river impacts (after dilution and dispersion). HAWRAT uses details of the highway catchment draining to the outfall, the flow rate of the receiving watercourse and its physical dimensions to calculate the available dilution of soluble pollutants using a mass balance approach. For the sediment-bound pollutants that cause chronic impact, the ability of the receiving watercourse to disperse sediments is considered and, if sediment is expected to accumulate, the potential extent of sediment coverage (the deposition index) is derived.
- Step 3, in river impacts post-mitigation to assess either the effectiveness of existing measures or the required scale of any proposed new measures (including retained existing measures).

The procedures involved in HAWRAT are summarised in Table 4.1. The adopted reporting method is based on a pass/fail response where 'Fail' indicates an unacceptable impact, a need to carry out further assessment steps, or a need to refer the situation to specialist judgement. 'Pass' indicates that there will be no short-term impact associated with road runoff. Long-term risks (using annual average concentrations) are needed to complete the risk assessment process. HAWRAT estimates in-river annual average concentrations for soluble pollutants (dissolved copper and dissolved zinc) due to road runoff and these can be compared with published Environmental Quality Standards (EQSs) to assess whether there is likely to be a long-term impact on ecology.

HAWRAT has been developed primarily for use on non-urban trunk roads and motorways in England and has been adapted to reflect conditions within Wales, Scotland and Northern Ireland. However, it has a number of limitations and is not directly applicable in some circumstances. Situations where HAWRAT may not be directly applicable include:

- urban highways (where a wider range of pollutants and larger concentrations may arise) may be under-represented by HAWRAT because of the reference datasets
- highways outside the UK (due to differences in rainfall, climate, vehicle fleet and other factors)
- highways where the receiving water course is tidal.

Table 4.1 Identification of the different stages involved in the Highways Agency Water Risk Assessment Tool (HAWRAT)

Stage of Assessment	Inputs	Outputs
Step 1 Runoff quality	• Traffic volume • Geographic location • 10 years of rainfall data, ~1000 rainfall events (embedded in HAWRAT)	• Runoff concentrations of soluble and sediment-bound pollutants for each event • Pass/Fail standards
Step 2 In river	• Outputs from Step 1 • Area draining to outfall • Characteristics of receiving watercourse	• Concentration of soluble pollutants for each event • Stream velocity at low flow • Deposition index (extent of sediment coverage) • Pass/Fail standards • Percentage settlement required to comply with deposition index • Average annual concentration of soluble pollutants
Step 3 After mitigation	• Outputs from Steps 1 and 2 • Existing and proposed mitigation measures −Treatment of soluble pollutants −Flow attenuation −Settlement of sediments	• Concentration of soluble pollutants after treatment • Concentration of soluble pollutants after further dilution • Pass/Fail standards • Annual average concentration of soluble pollutants after mitigation

4.1.3 IMPACT A Model to Assess the Environmental Impact of Construction and Repair Materials on Surface and Ground Waters

Through a project funded by the National Cooperative Highway Research Programme (NCHRP) researchers at Oregon State University, USA have identified potentially mobile constituents from highway construction and repair materials including those classified as conventional, recycled, or waste materials. As indicated by the project title 'Environmental Impact of Construction and Repair Materials on Surface and Ground Waters' (Project 25-09), the potential impact of these materials on surface water and groundwater was assessed but constituents originating from construction processes, vehicle operation, maintenance operations, and atmospheric deposition were not included. The outcomes from this research programme included:

- laboratory methods to realistically simulate the leaching of constituents from construction and repair materials in typical highway environments
- laboratory methods to evaluate the removal, reduction, and retardation of leached constituents from construction and repair materials
- extensive data sets of laboratory test results for highway construction and repair materials, expressed as both aquatic toxicity and chemical concentrations and

- a software program, IMPACT, that estimates the fate and transport of such leachates in the environment immediately surrounding the highway. The IMPACT software contains an extensive, readily accessible database of laboratory test results for materials ranging from common construction and repair products to waste and recycled materials proposed for use in highway construction (Nelson et al., 2001).

The developed mathematical model (IMPACT) is for use in the near highway (scale of metres) environment and consists of the fate and transport analyses related to removal, reduction and retardation processes, plus the generation of initial pollutant loadings. More specifically, the transport processes of advection and dispersion (in soil) are coupled to the removal, reduction and retardation processes of sorption, biodegradation, photolysis, and volatilization. Model output consists of flows, loads (mass), concentration of surrogate chemical (surrogate for toxicity), and toxicity of tested construction and repair materials in their appropriate reference environments. The model incorporates a database for all processed data from the project and therefore can be used to retrieve information such as EC50 and LC50 values, associated chemical concentrations, sorption data, and removal, reduction and retardation process parameters.

The model is written in Visual Basic for Applications for use in conjunction with the Microsoft® Excel 7.0 spreadsheet. This is a useful format for data storage and retrieval and provides ready accessibility. A full User's Guide (Hesse et al, 2000) is available, and further documentation is provided in the form of help screens within the model itself. User guidance is provided as well as input data for environmental parameters (e.g., soil properties, hydrologic parameters).

Leaching rates in the model are based on the flat plate results for the highway surface, piling, borehole, and culvert reference environments, while column leaching is considered appropriate for the fill reference environment. In addition, the model can simulate lateral movement away from the highway edge in a shallow aquifer. Hydrologic response to rainfall is simulated by using a runoff coefficient and surface roughness values characteristic of highway surfaces, and infiltration through a paved surface may optionally be determined based on empirical studies of cracks. Infiltration into soil (including the highway subbase) is based on soil properties such as hydraulic conductivity and porosity. While a constituent remains on the land surface, the process of volatilization and photolysis may degrade it. However, because the surface residence time of the highway runoff is typically very small, of the order of seconds in the absence of ponding, these two processes are considered to be unimportant degradation mechanisms. Biodegradation will continue along subsurface pathways, but is also unimportant for the important toxicity-causing metals, which do not biodegrade (or volatilise or photolyse).

Sorption is identified as the most important removal, reduction and retardation process. The model uses an explicit finite difference scheme to simulate the changes in concentration of the constituent as it migrates through the soil and is subject to advection, dispersion, sorption, and biodegradation (where relevant). As leaching rates decrease with time and cleaner water infiltrates through the soil, the desorption process is also simulated. Single-event or long-term (hourly or 15-min) rainfall data may be used to drive the fate and transport simulation. At the bottom of the soil profile, the model predicts flows, concentrations, and loads (product of concentration and flow) of the surrogate chemical, and associated toxicity.

The model was evaluated by comparing measured breakthrough curves (concentration versus time) from column studies with model predictions. Prediction of breakthrough of an organic constituent (2,4,6-trichlorophenol; TCP) was better than for breakthrough of the copper and zinc in ammoniacal copper zinc arsenate (ACZA) leachate. Breakthrough of ACZA arsenic was predicted reasonably well. The model results were typically conservative with breakthrough predicted to be no later than was observed in the laboratory column studies.

The IMPACT model suggests that most construction and repair materials behave in a benign fashion in the environment. On the highway surface, a combination of slow leaching, rapid transport and large dilution lead to very low initial concentrations of potential contaminants. Furthermore, rainfall is intermittent and leaching rates decrease with time. Transport in soils is generally very slow owing to high compaction and low hydraulic conductivities. Thus, vertical water movement in the highway base is slow, in the range of centimetres to metres per year. Vertical migration of contaminants is even slower because of the high sorption capacity of most soils. Model results showed vertical migration of contaminants on the order of only a few millimetres over a simulation period of many days. Hence, the model demonstrates the effectiveness of the near-highway environment in retaining the runoff generated from construction and repair materials.

4.1.4 A coupled MT–GA model for the prediction of highway runoff quality

Data-driven modelling (DDM) provides a methodology for predicting runoff pollutant concentrations as 'Grey Box' models of this type require only partial denotation of the underlying processes, while taking advantage of past events and available computing resources to deduce the likely outcomes of future events. Opher and Friedler (2009) have used the hybrid data-driven modelling methodology of MT–GA (model tree–genetic algorithm) as a predictive model for five common highway runoff pollutants. The selected pollutants represent those commonly found in highway stormwater runoff and cover three pollutant categories (heavy metals, organics and suspended matter). The target variables chosen for testing and demonstrating the proposed modelling approach were Pb_{total}, Cr_{total} and Zn_{total} (total=particulate+ dissolved fractions), TOC and TSS. TSS was selected for its significant positive correlation with many harmful pollutants found in highway runoff. These correlations make TSS an important goal for modellling, as it may serve as an indicator for other pollutants.

The derived models were trained and subsequently verified using a comprehensive data set of runoff events monitored for various highways in California, USA (68 runoff events monitored at 92 highway sites between 1998 and 2004). The five variables which were selected as potential inputs for the data-driven model were annual average daily traffic (AADT) [10^3 vehicles/d], antecedent dry period (ADP) [d], event rainfall [mm], maximum 5-minute rain intensity [mm/h] and antecedent event rainfall [mm]. Of these five explanatory variables, all possible sub-group combinations were examined.

The developed approach combines two data-driven methodologies, model trees (MTs) and a genetic algorithm (GA) in a coupled scheme of alternating execution. The GA searches for optimal model coefficients which are then incorporated by the MT into the tree structured model. The models consist of a set of deductive rules representative of an optimized model tree structure. The altered MT algorithm incorporates some knowledge of physical processes involved through non-linear equations. The resulting tree, therefore, contains nonlinear formulae at its leaves, unlike standard model trees, which carry a linear sub-model at each leaf. The correlation coefficients between predicted and observed pollutant EMCs are within the range of previously reported models of other modelling disciplines, or higher. As in the case of modelling methodologies based on regression analysis, a benefit of the MT–GA approach, apart from the estimation of pollutant concentrations in the runoff, is its emphasis on the most influential explaining variables, which appear to be ADP, AADT and a characteristic of the current storm (either rainfall volume or maximum rainfall intensity). ADP was identified as the only explaining variable which has a major role in explaining the variance of all five target pollutants. AADT is the second most common variable, taking part in all but the model for TOC. Rainfall and maximum rainfall intensity appear alternately in the models, one of the two in each one. The results obtained demonstrate that data-driven modelling is certainly worth further refinement and examination as a solution for highway runoff pollution modelling.

In those cases of relatively low model performance, such as for Cr_{total} and TSS, it is postulated that there may be other significant factors which have not been included in the modelling effort. The sensitivity of the model to changes in input values was low, while sensitivity to the incorporation of a certain attribute was very notable for all target variables. Each model is sensitive to a single attribute and quite indifferent to the other four. This one attribute is, in most cases that of daily traffic, with the exception of TOC, which responds to the presence of event rainfall rather than to traffic count.

Models of this sort may function as a tool for environmental risk assessment in seeking the best management practice (BMP) for handling highway runoff waters based on the modelled EMCs. When planning the route of a new highway or when assessing the need for runoff containment facilities alongside it, models of this type may be used with a few approximations of the yet unknown input data. Future storm characteristics such as total rainfall, maximum intensity or length of dry period must then be based on multi-annual data. As for all data-driven models, the performance is considerably less reliably when applied to input data which is out of the range of the training data used in the construction. Therefore applying these models to runoff-related data from geographical regions with climatic patterns significantly different than those found in California, would require a simple calibration process to achieve a better fit.

4.1.5 PREQUALE; A multiple regression approach for predicting Highway runoff quality in Portugal

This modelling approach has been developed based on data collected from six Portuguese road sites with a total of only 38 storm events. The development process was guided by the requirement that the model should predict at least two priority pollutants, should use simple input data, be easy to handle, should not require complex decisions during the calculations and should be based on a clear procedure that could be verified. PREQUALE consists of a multiple regression equation and its coefficients based on a set of five explanatory variables for the prediction of total suspended solids (TSS), Zn, Cu and Pb average concentrations in highway runoff (Barbosa, 2007). Although the predicted concentrations were very close to the observed concentrations it is accepted that the developed approach would benefit considerably from the support of a more extensive database incorporating road runoff characteristics.

Following a review of previously reported techniques (Driver and Tasker, 1990; Kayhanian et al., 2003; Hurtevent et al., 2006), appropriate selected variables were identified as ‘drainage area’, ‘percentage of impermeable area’ and ‘AADT’ to which were added ‘annual average rainfall’ and ‘mean storm event volume’. Using a multiple regression analysis, the objective was to identify the smallest set of these five independent variables which could best explain the selected pollutant concentrations. However, the final regression equation established for PREQUALE (shown below) surprisingly did not include the ‘AADT’ variable:

$$C = a_i (A^{\beta 1} \times I^{\beta 2} \times P^{\beta 3} \times P_{annual}{}^{\beta 4})$$

Where:
C (mg/L) = estimated site mean concentration of pollutant
a_1, β_1, β_2, β_3, β_4 = regression coefficients
P (mm) = mean annual volume of storm with duration of the concentration time and a return period of 2 years
A (km^2) = drainage area
I (%) = percentage imperviousness of drainage area
P_{annual} (mm) = annual average rainfall

PREQUALE produced very high R-square values for each of the tested pollutants with values of 0.9392 (TSS), 0.9648 (Zn), 0.9565 (Cu) and 0.9999 (Pb).

PREQUALE provides a preliminary method to enable the prediction of road runoff quality in Portugal which will assist in road runoff management and water resources protection. However, the tool has limitations, particularly in the limited data which has been used for its construction. It should also be noted that future changes in the use of roads, vehicles and engine construction as well as climatic changes could result in modifications to the current pollutant sources and loads.

4.1.6 The impact of AADT on highway runoff pollutant concentrations

Kayhanian et al. (2003) evaluated correlations between annual average daily traffic (AADT) and storm water runoff pollutant concentrations generated from California Department of Transportation (Caltrans) highway sites. The monitoring data includes event parameters such as precipitation (start and end time, maximum intensity, antecedent dry period), and runoff (total flow volume, peak flow rate, and start and end time).

Multiple linear regression (MLR) and analysis of covariance (ANCOVA) were used to address the impact of AADT on pollutant concentrations with thresholds for statistical significance set at a confidence level of 95% ($p<0.05$) for all analyses. The distributions of runoff quality data for each constituent were evaluated for approximate normality using normal cumulative probability plots of untransformed and log-transformed data. The transformation providing the best R^2 regression statistic was selected as the appropriate starting point for additional analyses. Distributions with R^2 values greater than 0.975 were considered adequately normal to meet the assumptions of subsequent analyses. The distributions of other continuous predictor variables (precipitation factors, antecedent conditions, AADT, and contributing drainage area) were also evaluated for approximate normality by inspection of cumulative probability plots, and were transformed to natural logarithms (event rainfall, maximum intensity, antecedent dry period, and drainage area) or cube-roots (cumulative precipitation), if appropriate.

MLR and ANCOVA methods were used to evaluate the effects of precipitation factors, antecedent conditions, AADT, contributing drainage area, and surrounding land use on highway runoff quality. MLR models were developed for each constituent. The primary assumptions of MLR analysis (equal variance and normality) were assessed by inspection of residual plots. Problems due to unequal variance and non-normality of residuals were largely avoided by transforming dependent and independent variables to approximate normality prior to analysis. Generally, all significant predictor variables ($p<0.05$) were included in the MLR model unless they exhibited symptoms of multi-collinearity or co-dependence in the set of predictors.

The final ‘‘optimized’’ MLR model was used to generate a new fitted variable calculated as the cumulative effects of the significant predictor variables for each constituent. This fitted variable was then included as the single covariate in the ANCOVA models used to evaluate the effects of surrounding land use. Because of imbalances in the representation of land use categories, interaction between individual covariates in the MLR model and the categorical variables could not be assessed in a statistically rigorous way. Instead, potential interaction effects were quantitatively evaluated by inspection of bivariate plots of the dependent variable versus the MLR-fitted data. In all cases, interaction was judged to be minimal and to have no substantial effect on the interpretation of the ANCOVA results.

The development of MLR-based runoff quality models has a number of practical applications. Two of the most important applications include estimating mass loads and using the model as a tool to address runoff management issues. The equations derived from multiple regression models can be used to estimate the pollutant event mean concentrations under different conditions. For example, the expected event mean total copper concentration in runoff from a specific (or predicted) storm event and location can be estimated as:

Total copper concentration µg/L = $e^{2.944 - (0.233X1) + (0.127X2) - (0.247X3) + (0.077X4) + (5.66X5)}$
where X_1 = Ln (event rainfall, cm);
X_2 = Ln (antecedent dry period, days)
X_3 = (cumulative precipitation,cm)$^{1/3}$
X_4 = Ln (drainage area, ha);
And X_5 = AADT x 10^{-6} (vehicles/day) .
The estimated EMCs can in turn be used to estimate the mass loading on a site-specific, regional, or watershed basis.

No simple linear correlations were found between highway runoff pollutant event mean concentrations (EMCs) and AADT, including for those pollutants that are known to be related to transportation activities (e.g. Pb, Cu, Zn, and oil and grease). However, AADT is not the only factor capable of influencing the accumulation and runoff of pollutants from highways. Other factors with significant effects include antecedent dry period, seasonal cumulative rainfall, total event rainfall and maximum rain intensity, drainage area, and land use. When the effects of these other factors were also considered, AADT was found to have a significant effect on concentrations of most constituents in highway runoff.

The effects of AADT, total event rainfall, seasonal cumulative rainfall, and antecedent dry period on pollutant concentrations in highway runoff were significant for more than 70% of constituents evaluated using multiple linear regression analysis. The effects of drainage area and maximum rainfall intensity were smaller and less frequently significant. AADT and other evaluated factors can be used as a practical tool for planning and prioritizing efforts for managing runoff quality in highly urbanized areas. Based on these results, contributing land use effects on runoff quality seem to be less consistent and less important than AADT and the other parameters evaluated in this paper. Consequently, land use characteristics may be less valuable in predicting runoff quality and in planning and prioritizing management activities.

4.2 Matrix assessment of available models
Table 4.2 presents an overview of the six tools described in Section 4.1 within a matrix format to support a comparative side-by-side evaluation of their purpose, data requirements and limitations etc. Consideration of the information summarised within the matrix indicates that predicting the impact of highway runoff on receiving waters is a fairly novel concept which, to-date, has only been considered in a relatively limited context. For example, whilst all six models are data driven i.e. based on local to national data sets, this does impact on their potential applicability within other geographic, climatic and traffic density/road use contexts. Whilst the model 'shell' or structure may be transferable, all models would ideally need to be re-calibrated using local data sets to ensure their fitness for purpose. In terms of the pollutants, the models are consistent with the literature on ecotoxicological impact (Sections 3.1 and 3.2) in that all six models include Cu and Zn within their target pollutants, as well as selected organic substances with TSS identified as both a pollutant in its own right as well as acting as a surrogate for a wider range of particulate associated substances. A key substance identified in the literature that does not appear to be specifically addressed within any of the models is NaCl, and inclusion of this parameter within existing approaches is highlighted as research priority.

Whilst all six models predict pollutant loads within an aquatic phase (either within the highway discharge or, more comprehensively, the receiving water), only two of the models (HAWRAT and IMPACT) consider any further receiving compartment e.g. groundwater or sediments. Inclusion of a methodology to consider the impacts of highway discharges on groundwater is a more recent addition to HAWRAT and is based on a SPR approach as recommended (Method C) in the Highways Agency Road Drainage and the Environment Manual (2009). The methodology is now used quite extensively in UK drainage infrastructure assessment. The model IMPACT considers the impact of substances leaching from various highway construction materials in terms of their potential to migrate vertically and horizontally to shallow groundwaters. Whilst only a relatively small number of material types are tested, the process based approach utilised could be further developed / inform a model focusing on the impact of pollutants mobilised by highway runoff. In identifying whether an impact is likely, predicted concentrations are then compared within pertinent receiving water quality standards as part of the model (e.g. HAWRAT) or manually by the user (e.g. PREQUALE). Whilst comparison of predicted concentrations with, for example, EU environmental quality standards, does imply consideration of receiving water ecological impacts, none of the models specifically incorporate a biotic component.

Given the importance of groundwater resources as drinking water resources and the fact that complete source–pathway-receptor chains from highways to groundwater have been widely – if not routinely – reported, the absence of a robust tool to predict when impacts will occur is identified as a key knowledge gap. In terms of predicting surface water impacts, the inclusion of a biotic component would be a welcome addition to the approaches developed to-date. Whilst the utility and purpose of comparing predicted values with EQS is acknowledged, research has shown that exceeding EQS does not always equate with an ecological impact (and vice-versa). The concept of runoff specific thresholds is included within HAWRAT but is, to-date, of limited regulatory interest as the focus of compliance remains (at least within EU Member States) on EQS. EU Biotic EQS are now available for some pollutants (e.g. for mercury and fluroanthene; EU Priority Substances Directive, 2013) and it is hoped that this will act as an incentive for the development of impact prediction models which incorporate one or more biotic groups.

Table 4.2. Matrix overview of tools developed to predict the impact of activities on receiving waters

	HAWRAT	**SELDM**	**IMPACT**	**MT-GA**	**PREQUALE**	**Impact of AADT**
Developers	Environment Agency and UK Highways Agency	US Geological Survey and Federal Highway Administration	National Cooperative Highway Research Programme and Oregon University	Opher and Friedler (2009)	Barbosa (2007)	Kayhanian et al., 2003
Country	UK	USA	USA	USA	Portugal	USA
Purpose	Predict impact of highway discharges - with and without treatment - on receiving water	Estimate the level of risk to receiving waters, need for mitigation measures and their impact	Estimate the transport and fate of soluble pollutants released from highway materials	Predict runoff concentrations of highway pollutants	Predict highway runoff pollutant concentrations.	Estimate EMCs and mass loads for a storm event of a predicted magnitude
Pollutants	Acute impacts: dissolved Cu and Zn EMCs; Chronic impact: event mean sediment concentrations for total Cu, Zn, Cd, pyrene, fluoranthene, anthracene, phenanthrene and total PAHs	Range of organic and inorganic pollutants	e.g. 2,4,6-trichlorophenol; TCP); ammoniacal copper zinc arsenate (ACZA) leachate (chemical concentrations and aquatic toxicity data)	Total Pb, Cr_l and Zn, TOC and TSS.	TSS, Zn, Cu and Pb	Pb, Cu, Zn, and oil and grease
Input variables used / considered	Stage 1: site traffic data and geographic location Stage 2: catchments size and receiving water characteristics Stage 3: Suds data sets (e.g. CIRIA, 2015)	Site and receiving water characteristics; precipitation data, water quality, and mitigation performance data derived from national datasets for local land-use types	Road material type; soil type; hydrological parameters.	AADT; ADP; event rainfall; max. 5 minute rain intensity, antecedent event rainfall	drainage area; % imperviousness; AADT; annual average rainfall; mean storm event volume	Event rainfall, ADP, cumulative precipitation, drainage areas, AADT
Methodology	Stochastic: draws on national data sets to predict runoff quality linked to AADT, geography; catchments size, receiving water characteristics and suds data to assess removal of soluble pollutants and settlement of sediments	Stochastic; empirical model based on national data sets and statistics rather than physiochemical equations.	Deterministic; processes modelled include advection, dispersal, sorption, biodegradation, volatilisation and photolysis	Stochastic; combination of model tree – genetic algorithm data driven modelling approaches; regression analysis	Stochastic; multiple regression to predict pollutant concentration	Stochastic; multiple linear regression to predict pollutant concentration
Receiving compartments	Focus on surface water but includes assessment methods for groundwater discharges and spillage risk assessment	Surface water	Surface water and groundwater	Highway runoff	Highway runoff	Highway runoff
Limitations	Urban highways (where a wider range of pollutants and larger concentrations may arise); highways outside the UK (due to differences in rainfall, climate, vehicle fleet and other factors); highways where the receiving water course is tidal.	Limited data sets generates substantial uncertainties associated	Few highway materials tested	Based on Californian data/climatic conditions - require calibration for use in other contexts.	Based on data from 6 sites and 38 events	Based on Californian data/climatic conditions - require calibration for use in other contexts.

5. Conclusions and recommendations for further work

The studies reported in this review relate to data collected from throughout the EU and also internationally. Therefore, they pertain not only to a wide variety of climatic and geographic circumstances but have also involved a range of sampling and analytical protocols as well as experimental designs and test species. In addition, many of the studies do not provide full details of the geological, physico-chemical and/or hydrology of the receiving water during sample collection, with the occurrence or magnitude of any seasonal variations in identified parameters even less well reported. For these reasons it is difficult to integrate identified data sets with any degree of confidence. However, despite this, it is recognised that end-users e.g. National Road Administrations and environmental protection agencies, are required to make decisions now on when, where and how road runoff should be treated. Within this pragmatic context and in respect of the substantial number of papers reviewed, the following sections provide a brief overview of the evidence base associated with each of the following key questions:

- Does highway runoff impact on the ecological and/or chemical status of receiving waters?
- What sort of impacts have been reported?
- Is there a relationship between AADT and ecological impact?
- What are the key contaminants in highway runoff?

The section then concludes with a discussion on the implications of this review for defining receiving water vulnerability within the CEDR PROPER project and develops a series of recommendations for further work.

Does highway runoff impact on the ecological and/or chemical status of receiving waters?

Using a ‘weight of evidence’ approach, it can be concluded that highway runoff can impact negatively on the chemical and ecological status of receiving waters. Whilst there are studies where the ecology of surface waters downstream of a highway discharge site does not significantly differ from those upstream of the discharge point, several detailed studies have reported a change in the nature or composition of downstream ecologies. For example, differences in species composition, abundance and feeding behaviour have all been identified as arising from the discharge of road runoff to a receiving water body. However, the challenge of establishing a causal relationship between the chemical and ecological statuses remains as even runoff discharges identified as exceeding certain EQS are not consistently associated with poorer ecological status. Hence much work remains to be done in relation to understanding the impact and interactions of pollution mixtures on receiving water biota (at both acute and chronic levels) before levels of risk arising from highway discharges can be robustly characterised.

What sort of impacts have been reported?

Both acute and chronic impacts have been reported in the field and in laboratory experiments. Acute impacts have been fairly extensively considered and are detected through the use of 6-24 hour toxicity tests which typically focus on endpoints such as mortality, effect and inhibition with chronic effects determined through either undertaking repeated exposure tests or monitoring test species for an extended time period (e.g. six weeks) to identify the occurrence of any lag effects. However, few long term studies exist which investigate the impact over extended time scales of several years or even decades. With regard to impacts on vertebrates, research has been undertaken on the impacts of highway runoff using a variety of fish species (at various life stages) and test types (from impact on gene expression to mortality). Impacts on a range of species (e.g. brown trout, roach, bullhead and stickleback) have been reported. Results show that different species vary in their sensitivity to particular pollutants e.g. Hurle et al. (2006) demonstrated the common minnow is sensitive to Cu and Zn whereas the bullhead was not. Several studies have indicated fish are more sensitive to Cu than Zn but impacts have also been associated with the exposure of various fish species to PAHs and suspended solids. In contrast, some studies involving exposure of fish to highway discharges have not detected any impact e.g. Bruen et al. (2006). In attempting to explain these variations in results, researchers draw attention to a range of factors, with particular emphasis placed on *in situ* water hardness; several studies have shown increasing toxicity with decreasing water hardness levels. In addition to fish studies, a limited number of studies have looked at the impact of highway discharges on frogs. Differences in metal uptake processes by life stage have been observed with metals only able to accumulate passively during the 'egg stage' whereas in the 'tadpole-stage' both active and passive uptake of metals occurs through the ingestion of food, respiration through gills and passive diffusion through the body surface (Meland et al., 2013). A further study compared the accumulation of PAHs in tadpoles, dragonflies and plants collected from three sedimentation ponds, reporting that levels accumulated by tadpoles were an order of magnitude higher than those accumulated by dragonflies, but were considerably lower than those observed in plants (Grung et al., 2016).

A considerable body of literature has focused on assessing the impacts of highway discharges on aquatic invertebrates in receiving water and sediment compartments. Both laboratory and field experiments have been undertaken which, taken together, indicate that receiving waters downstream of highway discharges often report lower levels of diversity and abundance using a variety of methods and classification indices. However, whilst identifying differences, changes detected were not always at a statistically significant level with other studies reporting metal concentrations in sediment and water being only marginally correlated with the metal body burdens of the sampled invertebrates (Meland et al., 2013). As with fish, different species demonstrate differing behaviours/levels of sensitivity as a result of exposure to highway runoff. For example, the abundance of mosquitoes (e.g. *Paratanytarsus grimmii*,) significantly increased with increasing concentrations of sediment contaminants whereas the occurrence of the aquatic insect *Tanytarsus fuscithorax* significantly declined with increasing concentrations of zinc in surface waters due to leaching from sediments (Pettigrove et al., 2007). Whilst road construction and operation does impact on local ecosystems in variety of ways, research by Englert et al. (2015) indicated that the main driver in ecological changes was alterations in water quality as a result of highway discharges rather than any associated morphological modifications.

With a focus on groundwater impacts, the quality status of groundwater is an important issue for all EU Member States as it is a major source of public water in many areas. Hence the quantity and quality of groundwater have been the subject of considerable research, with a focus on any practices that may jeopardize the chemical status. The receipt and slow infiltration of highway runoff to recharge local groundwaters via sustainable drainage systems (e.g. swales and infiltration trenches) is a common practice, and thus the potential for groundwater to receive highway flows with elevated concentrations of a range of pollutants exists. Interrogating this hypothesis has been the subject of several research studies, with regard to both direct (infiltration of runoff via SUDS) and indirect (leaching of pollutants from soils) processes.

With regard to the impact of direct infiltration of runoff on groundwater, results of samples collected from a 30 year old infiltration basin in Lyon, France, indicated little mobilisation or downward transfer of pollutants from contaminated basal sediments (Datry et al., 2004). Whilst other studies have reported similar findings, other researchers note the potential for the drying out and 'cracking' of basal sediments during dry weather which may lead to preferential flow paths to greater depths during storm events (Winiarski et al., 2006). In recognition of this event-driven occurrence, the importance of regular maintenance practice (and sediment removal) to prevent potential contamination of groundwater sources is recommended. Recommendations on the maximum background metal concentrations in underlying infiltration soil layers have also been developed, as metal displacement may occur as a result of competitive adsorption/exchange and/or dissolution effects posed by the multi-component system (Pitt et al., 1999). In contrast to metals, several studies have concluded that de-icing materials are a particular threat to the good chemical status of groundwaters. For example, a study in Toronto, Canada, reported that highways annually received more than 70,000 tonnes of NaCl road de-icing chemicals of which as much as 50% – 60% of the applied chloride entered the shallow unsaturated zone (Howard and Haynes, 1993). However, research undertaken by TRL (2002) reported that such high winter loadings can be rapidly reduced by dilution in the unsaturated zone to below threshold levels during summer and autumn periods. With regard to indirect processes, several studies have reported that metal depositions within roadside soils reduce to background concentrations within approximately 10 m of the road edge (e.g. Ward, 1990), with the majority of studies concluding that soil pollutant concentrations also rapidly decline with increasing depth suggesting any horizontal or vertical migration of metals is limited. The only exception to this general trend is for NaCl concentrations which can remain elevated for greater distances from the road edge and at greater depths within soil profiles (TRL, 2002) leading to concerns over the potential for chlorides to facilitate the movement of previously adsorbed metal species.

What are the key contaminants in highway runoff?

The reviewed studies have identified a range of factors which contribute to the ecological impacts reported in receiving water compartments. Because test species and experimental methodologies utilised as well as receiving water characteristics showed considerable variation, it is not yet possible to draw robust conclusions on particular pollutants or pollutant groups responsible for reported impacts. However, it is possible to highlight certain trends that seem to be emerging from the prolific number of studies available within the literature. For example, the majority of fish studies identify Zn and particularly Cu as being toxic to several (although not all) fish species, with a key variable being the inverse relationship between water hardness and ecotoxic response (e.g. Hurle et al., 2006). The impact of NaCl de-icing salts appears to vary greatly on a species by species basis with toxic effects (where they occur) commencing at concentrations that range by over an order of magnitude. NaCl de-icing materials are also identified as the parameter of most concern with regard to prejudicing the chemical status of groundwater bodies. With regard to impacts on aquatic invertebrates, several studies have identified PAHs as of particular concern, with studies of gammarids identifying the pollution group as responsible for most of the toxicity reported (e.g. Maltby et al., 1995b). From the literature, it is also possible to make some general statements with regard to the temporal nature of the delivery of pollutant loads. For example, several studies have shown samples collected from the start of the event are generally the most toxic with typically more than 40% of the toxicity being associated with the first 20% of discharged runoff volume and 90% of the toxicity occurring during the first 30% of storm duration (Kayhanian et al., 2008). Similar results, demonstrating a sharp decline in runoff toxicity through individual storms have been reported by Mayer et al. (2011) with Perikaki and Mason (1999) expressing concern over the impact of ‘first flush’ events on receiving bodies particularly under low flow conditions.

Is there a relationship between AADT and ecological impact?

The annual average daily traffic (AADT) density is often used as an influencing parameter to identify when runoff from a highway should be treated prior to discharge to a receiving water. For example, Meland (2016) identifies 10,000–15,000 vehicles/day as the critical limit above which runoff treatment is required. Likewise the German Water Association technical standards for highway runoff identify an AADT of 15,000 as the level at which highway runoff should be treated prior to discharge (ATV, 2002). However, whilst some studies do show an association between AADT and ecotoxicolgical response (e.g. Mayer et al., 2011), available data do not indicate a consistent linear relationship between the number of vehicles and pollution concentrations in highway runoff. Therefore, whilst linking AADT to impact appears intuitive, a multiplicity of factors (pollution mixture effects, receiving water characteristics, climate and geology) are also routinely identified as enhancing (or mitigating) receiving water body impacts. In addition, the construction and operation of roads have also been identified as influencing local ecosystems through inducing changes in aspects not directly associated with runoff such as changes in soil density, temperature, soil water content, and light levels. Little is known about the impact or interplay of these factors on receiving environments, either in combination with runoff discharges or during dry weather. Researchers have also drawn attention to the need to consider other sources of diffuse pollution in both urban (e.g. CSOs) and rural (e.g. agricultural runoff) catchments, making the separate identification of highway runoff impacts difficult. The use of AADT as a predictor of ecological impact is therefore best used as a guiding principle only, until further research on the processes and interactions governing ecological responses of receiving waters to highway runoff is undertaken.

Implications for defining receiving water vulnerability within the CEDR PROPER project
In developing this review of current knowledge on the vulnerability of European surface water and groundwater to road related pollution, it is clear that the term 'vulnerable receiving water' is used in a variety of contexts leading to different definitions and interpretations. For example, the term has been used in relation to a number of bio-phys-chemical aspects including:

- Hydrological aspects: e.g. surface waters may be especially vulnerable to highway runoff during low-flow conditions
- Chemical aspects: e.g. highway runoff poses a greater risk to 'soft water' receiving waterbodies with regard to metal pollution
- Ecological aspects: e.g. water bodies that are host to sensitive species e.g. salmon fisheries are particularly vulnerable to highway discharges
- Geological aspects: e.g. chalk aquifers and shallow aquifers are often described as being vulnerable to highway discharges, especially following de-icing operations

The term vulnerable is also used in relation to a range of anthropogenic factors which can impact on receiving waters such as

- Actual / planned use of the water body: e.g. water bodies identified as sources of drinking water are vulnerable and hence given legislative protection
- Surrounding land use: receiving waters in urban areas are vulnerable to not only highway runoff but to discharges from combined sewer overflows (CSOs) and aerial inputs whereas rural receiving waters are additionally vulnerable to inputs from agricultural runoff
- Traffic density / road lay out/ driving styles: have all been identified as influencing the vulnerability of a receiving water body

The final task of WP2 is to develop a decision-support tool to assess receiving water vulnerability to highway traffic pollution as part of a suite of measures to enable NRAs to minimise the impact of traffic-related pollution on receiving waters within a risk assessment and management framework. It is therefore particularly important to develop and agree a common definition of the term 'vulnerable water body' for use within the CEDR PROPER project. The various definitions which have been used can be broadly categorised into the inherent characteristics of the receiving water (i.e. its hydrology, hydrogeology and chemical and ecological status) and the influence of anthropogenic activities on their functioning e.g. road construction and operation and actual/planned use as a resource / disposal body. Within a risk assessment and management context, the conventional approach to assessing the level of risk generally involves three key stages:

- Identification of hazards
- Consideration of the likelihood of the identified hazards impacting on receptors
- Consideration of the magnitude of impact if hazards impact on receptors

Within a highway discharge risk assessment and management context, and based on the literature reviewed in this deliverable, Table 4.3 suggests the following definitions of key terms for use within future CEDR PROPER outputs.

Table 4.3 Definitions of key risk assessment terms within the CEDR PROPER project

Term	Descriptor
Risk	The likelihood that a specified event (the discharge of highway runoff) will negatively impact on the status of a receiving surface water or groundwater body
Hazard	Highway discharge and dispersion
Likelihood of occurrence	Benchmarking of EMC data against pre-developed scales which define high, medium and low levels of pollution concentrations/loads
Magnitude of impact	Informed by an assessment of the inherent characteristics of the receiving water (i.e. an integration of hydrology, hydrogeology and chemical and ecological status data sets benchmarked against pre-developed scales which define high, medium and low levels of receiving water vulnerability)

Recommendations for further work

Within this review of current knowledge on the vulnerability of European surface waters and groundwater to highway discharges, several areas have been highlighted as requiring further work. These evidence based knowledge gaps are summarised below, with a view to both informing future CEDR PROPER deliverables and future CEDR research agendas.

- There is a clear need to better characterise the link between highway runoff discharges and any subsequent ecological effects in receiving rivers/streams. In designing and undertaking new research, studies which integrate both population / community techniques and analytical chemistry determinations should be prioritised.
- Whilst acute ecological impacts are well described, relatively little is understood of the response of receiving water ecologies to repeated, low level (chronic) exposure to highway drainage
- Very little work exists to quantify the relative contribution of dry/wet aerial deposition to highway runoff pollutant loadings. In addition, the significance of 'on-highway' processes for the determination of final highway discharge outputs to the receiving waterbody remains an unresolved issue.
- Development of a harmonised pan-European approach to monitoring the state of urban/highway surfaces, surface water and groundwater (and the specific urban and transport pressures placed upon them in both acute and chronic terms) would underpin the development of comparable data sets.
- Elaboration of definitions of vulnerable surface water and groundwater that can be potentially applied at a European-wide scale. Identify criteria and supporting indicators to inform development of a vulnerability classification framework and source data sets to inform the application of benchmarking schemes.

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