D2.2 Construction, operation and maintenance of roads: parameters to assess surface and ground water vulnerabilities and associated risks

CEDR PROPER PROJECT

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Executive summary

This report is the second 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. Drawing on the findings of D1.1 (Pollutant concentrations and loads discharging from roads) and D2.1 (Vulnerability of European surface waters and groundwaters to road related pollution), its focus is the development a justified list of parameters which can be used to assess the vulnerability of receiving waters to highway traffic pollution. An assessment of environmental legal requirements and constraints relating to road construction, operation and maintenance activities is provided in D2.3 (An evaluation of International, European and national legislative frameworks and approaches).

The report begins with a definition of key terms and uses the three stage risk assessment approach proposed in D2.1 to frame subsequent sections. These include the identification of parameters to inform an assessment of the likelihood of contaminated highway runoff loads occurring (Section 2) and the magnitude of impact of highway runoff within receiving surface and ground waters (Section 3). As well as listing the parameters themselves, the nature and type of influence of each parameter is described and an indication is provided of how changes in the parameter can increase - or decrease - highway pollution loadings (i.e. likelihood of occurrence) and vulnerability of receiving waters (i.e. magnitude of impact).

Parameters identified in this report essentially form a 'long list' from which the criteria and supporting indicators for use within Task 2.5 (Development of a decision-support tool to assess receiving water vulnerability to highway traffic pollution) will be selected. This short-listing process (described in Section 4) will involve a screening of all parameters in relation to a range of aspects including feasibility, scientific soundness, mutual independence, operationality and to remove redundancy. The results of this initial parameter screening approach will be discussed with IAB members and their comments/preferences taken into account in the final criteria/indicator selection process.

Acknowledgements
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1. Introduction

This report draws on the findings of reviews of pollutant concentrations and loads discharging from roads (Deliverable 1.1 of the PROPER project) and the vulnerability of European surface waters and groundwaters to road related pollution (Deliverable 2.1 of the PROPER project) to develop a justified list of parameters to support an assessment of the vulnerability of receiving waters to road-related pollution activities. Together with the legal requirements/constraints identified in D2.3 (An evaluation of International, European and national legislative frameworks and approaches), parameters identified in this report essentially form a 'long list' from which the criteria and supporting indicators for use within Task 2.5 (Development of a decision-support tool to assess receiving water vulnerability to highway traffic pollution; Deliverable 2.5) will be identified. The process of short-listing parameters from the long-list developed within this report will take place within T2.5 and involve discussions with IAB members (commencing in October 2018; Month 14 of the PROPER project). Within this report, criteria are defined as major established factors on which the final judgement, evaluation or decision is made i.e. the principal areas of concern in the decision-making process (Ellis *et al.*, 2008). Indicators refer to the diagnostic states or conditions that describe relevant and appropriate properties of the given criteria. Benchmarking (at a criterion or indicator level) is the process of allocating scores to the aspects or options being assessed using either quantitative – or where not available - qualitative data (Ellis *et al.*, 2008).

As this deliverable effectively distils findings of D1.1 and D2.1 into a format to be used in the development of a decision-support tool (DST), it is pertinent to briefly summarise the requirements here. The aim of the Deliverable 2.5 DST is to enable National Road Administrations (NRAs) to assess receiving water vulnerability to highway traffic pollution as part of a suite of measures to minimise the impact of traffic-related pollution on receiving waters within a risk assessment framework. As the focus of the D2.5 DST is the assessment of receiving water vulnerability, it is important to develop and agree a common definition of the term 'vulnerable water body' for use throughout the CEDR PROPER project. A review of D1.1 and D2.1 indicates that the term is used in a variety of contexts leading to different definitions and interpretations which can be broadly categorised into:

- inherent bio-physico-chemical characteristics of the receiving water (i.e. its hydrology, hydrogeology, chemical and ecological status)
- influence of anthropogenic activities on the functioning of a water body e.g. road construction and operation and actual/planned use as a resource / disposal body.

The conclusions of D2.1 included the application of a conventional three stage risk assessment approach (i.e. identification of hazard, its likelihood of occurrence and magnitude of impact) within a highway discharge context, leading to the following definitions of key risk assessment (RA) terms for use within this – and other – CEDR PROPER outputs (see Table 1.1). Application of this RA approach essentially discriminates between the influence of anthropogenic activities on receiving waters (used to inform an assessment of likelihood of the defined hazard occurring) and the inherent bio-physico-chemical characteristics of the receiving water (used to inform benchmarking of the resulting magnitude of impact).

Table 1.1 Definitions of key risk assessment terms within the CEDR PROPER project (Revitt et al., 2018)

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 event mean concentration 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)

Within this context, this deliverable (D2.2) sets out a long list of parameters to support an assessment of surface and ground water vulnerabilities and associated risks due to the construction, operation and maintenance of roads. Parameters to enable the influence of road construction, operation and maintenance (i.e. anthropogenic aspects) are drawn from D1.1 (Fernandes and Barbosa, 2018; see Section 2 of this report) and parameters addressing the inherent bio-physico-chemical aspects of receiving waters are derived from D2.1 (Revit et al., 2018; see Section 3 of this report).

2. Identification of parameters to inform an assessment of the likelihood of contaminated highway runoff loads occurring

2.1 Identification of characteristics that influence road runoff contaminant concentrations during road operation

The evaluation of the impacts caused by road runoff to receiving water bodies remains a complex issue that needs to integrate a range of aspects including the variability of discharge volume and quality, as well as the discharge location and characteristics of the receiving water body (Barbosa et al, 2011). Data gathered within several monitoring studies, as well as field evaluations, carried out internationally enable the primary factors influencing road runoff contaminants to be identified. For an overview of the environmental legal requirements and constraints relating to road construction, operation and maintenance activities see Deliverable 2.3 (An evaluation of International, European and national legislative frameworks and approaches).

The key road runoff contaminants and their sources were identified in Deliverable 1.1 of the PROPER project (Fernando and Barbosa, 2018). This included identification of the key factors that may influence pollutant concentration and loads and their categorisation into three broad groups of factors:

- road and site characteristics
- climate conditions;
- physical and chemical parameters

Aspects related to rainfall pattern, road characteristics, ambient conditions and environmental attributes are reported to have simultaneous - and sometimes contradicting influences - on the magnitude of pollutant loads in road runoff (Opher and Friedler, 2010). The cause and effect relationships with regard to pollutants in road runoff are complex and data indicates that they cannot be universally established. However, there is general agreement on what these process are. These difficulties are reflected in the road runoff concentration prediction tools (see Deliverable 1.2) that include different variables depending on the studies used to inform their development and demonstrates the variations in pollutant loads monitored at specific site, regional or country-level. For example, despite the widely reported increase in pollutant wash-off with increasing rainfall intensity, several authors also report negative correlations between pollutant concentrations and rainfall intensity. These statements mean that, as in any other field of knowledge, the evaluation of relevant factors must be assessed carefully and preferably by someone who is acquainted with the phenomena.

The main characteristics that influence road runoff concentrations and loads discharging during road operation phase are summarised in Table 2.1. As the influence of these factors can change, according to the circumstances, the Table contains a column identified as '*Influence*' where positive indicates a contribution to higher pollutant concentrations and negative indicates the generation of lower pollutant loads. For example, the road and site characteristics and the type of pavement (i.e. its porosity) influence road runoff water quantity and quality as the higher the porosity, the higher the runoff infiltration and pollutant retention within the pavement structure. However, results from various research projects are not always consistent. Research by Legret (2001) found porous pavements removal rates, of between 20 and 75% for metals, and more than 85% for TSS. These removal rates are mostly due to the retention of the fine particles by the porous pavement. In contrast, Driscoll et al. (1990) did not observe any influence of the types of road surface material on runoff quality and pollutant loads. The deterioration of the pavement material will have influence on the porosity and filtration effects, therefore the process rates change during the lifetime of the pavement and according also to maintenance activities. A score is allocated to each impact identified in Table 2.1 on a scale of -5 to 5. A positive score indicates the parameter has the potential to increase pollutant concentrations and a negative score signifies the parameter can have the potential to reduce pollutant concentrations. The magnitude of the scores are allocated on a relative basis, where +5 is relatively the greatest potential to increase pollutant concentrations and -5 relatively the greatest potential to decrease pollutant concentrations. However, it does not mean that the net result follows all scores, as several site specific conditions can influence the direction of impact (as indicated by some scores being allocated both a negative and positive value).

Table 2.1 Characteristics influencing runoff quality during road operation (Opher and Friedler, 2010; Trenouth and Gharabaghi, 2016)

Type of factors	Factors	Influence (positive or negative)	Observation	Score
Road and site characteristics	Traffic (e.g. Annual average daily traffic; AADT)	**Positive:** a major sources of pollutants, increasing traffic loads will contribute to the accumulation of pollutants on the road surface **Negative:** turbulence caused by vehicles will remove pollutants from the road lanes	In most cases traffic is closely correlated with adjacent land use (i.e. more traffic occurs in urban areas making it difficult to distinguish between the effect of traffic and of the land-use on the pollutant concentrations). Positive correlation between AADT and concentrations of TSS and heavy metals	+/-3
	Area of the drainage basin	**Positive:** increase in particulate-associated constituents **Negative:** decrease the concentrations of dissolved constituents	Low influence on road runoff concentrations. Dependent on % of the total catchment area occupied by road pavement	+/-1
	Type and condition of pavement (e.g. age, permeability)	**Negative:** Quality of road runoff is improved by passage through porous pavement.	Different permeability affects volume and pollutant loads of runoff. Effect of material (e.g. asphalt, concrete) on road runoff loads appears to be minimal. Pavement deterioration could either increase (by contributing decomposition products) or reduce loads (by filtration in the road base materials)	-2
	Vehicle and driving characteristics	Vehicle and driving characteristics (e.g. speed and braking frequency) may influence the general effect on road runoff loads. **Positive:** increase of exhaust emissions and mechanical wear.		+3
	Land-use and road structures		A large part of nutrients washed off by runoff originates from atmospheric deposition in agricultural areas.	-
Climate conditions	Rainfall intensity	**Positive:** Rainfall intensity plays an important role in the detachment and mobilisation of pollutants. High rainfall intensity contribute to mobilization of pollutants associated with particles		+3
	Volume of precipitation	**Positive:** Increase the total load that is washed off the surface **Negative:** As the mass of pollutants in a road surface is limited, increasing rainfall volume leads to a decrease of the concentrations due to dilution effects	In some cases rainfall itself can be a source of pollutants	+/-2
	Antecedent dry period	**Positive:** during ADP, pollutant accumulate in the road surface. The pollutant build-up process is not linear as it increases quickly after a rain event, slows down after some time and then reaches a maximum. **Negative:** On the other hand, long ADP also promotes pollutant dispersion and degradation through other mechanisms than rainfall	ADPs longer than a certain threshold have an identical effect in the pollutant load available for wash-off by a rainfall event.	+/-3
	Temperature increase, humidity, air/ winds and sunlight	**Negative:** volatilisations and oxidation processes		-3

Vehicle and driving characteristics influence pollutant sources through exhaust gases, tyre wear, fuels, oils, lubricants and greases (e.g. impact of the changes in the fuel composition on the concentration of lead in road runoff). Traffic conditions and type and material of road structures (e.g. galvanized bridges, drains) can be important contributors of metals to runoff whereas road conditions (i.e. age and composition of pavement) can be important contributors of PAHs.

Rainfall intensity and volume and antecedent dry period are relevant driving variables for road runoff pollution content (Higgins, 2007; SETRA, 2007; Barbosa et al., 2011). Besides the rainfall pattern, climate/weather variables such as the temperature and solar radiation may have some influence, according to the site/region/country. In Portugal, for instance, a few measurements of PAH and oil and grease revealed values below the quantification limit (Barbosa et al. 2011). The effect of the temperature and the high solar radiation, enhancing the volatilisation and degradation of the pollutants were likely to explain these observations. This is in line with Crabtree *et al.* (2008) who identified the presence of PAH only in the colder regions of England.

2.2 Identification of characteristics that influence road runoff contaminant concentrations during road construction and maintenance

Road construction is an activity that results in land use change. The magnitude of the change and construction works are informed by the road project and road design. Naturally, a multilane highway built in a natural and hilly landscape will result in higher site alterations. It is common to have the removal of vegetation followed by moving the soil and changing the surface topography, both by excavating or creating embankments, in order to implement the base foundation where the road platform will be constructed. In some cases, the site's geology can be impacted by the use of explosives or machinery to create tunnels (open or closed). Following the direct impact on the site landscape, soil, geology, vegetation and fauna, there may be indirect impacts on all water bodies. Aquifer structure may be impacted by altering the soil and local geological structure. On the other hand, when it rains during the construction stage there is a decrease of water infiltration and an increase in surface runoff, with erosion processes taking place and increased transportation of sediments (and organic matter) to receiving waters.

Whilst during road construction there is no traffic associated with the road in the conventional sense, heavy machines and trucks are in operation. They can be a source of oil and grease leakages, as well as fuel – in particular if local fuel deposits are created and on site refuel operations take place. Also common is the installation of building yards in the vicinity of the road works to allow storing of machinery, materials and even the provision of sanitation, canteen and dormitory facilities for road construction workers. All these different activities may produce contaminants which have the potential to impact on water bodies. Road maintenance involves similar activities to road construction, but to a lower extent and degree. The characteristics that influence road runoff loads during the construction and maintenance phases are summarised in Table 2.2a and Table 2.2 b, respectively.

Table 2.2a Characteristics that influence road runoff contaminant concentrations during road construction

Characteristics	Water indicator contaminants	Observations	Score
Area of land disturbed during construction	TSS	Recommendations show be made to minimize impacts on water bodies and systems. Vegetation removal and soil/rock moving are activities that should take place during the dry months to prevent erosion and sediments release.	4
Total number of workers	Faecal coliforms		3
Number of wet months during which soil and vegetation were moved	TSS		5
Protected receiving water systems or protected areas in the catchment	TSS, oil and grease, faecal coliforms, total hydrocarbons	Usually there are recommendations on how to perform these activities in order to minimize impacts on water bodies and systems	4
Heavy machinery stored and being subject of maintenance at the construction / building site	Oil and grease, total hydrocarbons	Major issue	4
Fuel tanks at the site	Total hydrocarbons, metals		2

Table 2.2b Characteristics that influence road runoff contaminant concentrations during road maintenance phases

Characteristics	Water indicator contaminants	Observations	Score
Area of land disturbed during maintenance	TSS	Recommendations show be made to minimize impacts on water bodies and systems. Vegetation removal and soil/rock moving are activities that should take place during the dry months to prevent erosion and sediments release.	4
Total number of workers	Faecal coliforms		3
Number of wet months during which soil and vegetation were moved	TSS		5
Heavy machinery stored and being subject of maintenance at the construction / building site	Oil and grease, total hydrocarbons	Major issue	4

3. Identification of parameters to inform an assessment of the magnitude of impact of highway runoff on receiving waters

3.1 Receiving surface waters

3.1.1 Introduction

As part of a Portuguese/Slovenian collaboration, Brenčič et al. (2012) have proposed a decision tree approach for the identification of water bodies in their home countries which are sensitive to highway runoff pollution. Three categories were identified as: ‘sensitive’ (where the direct discharge of road runoff to water bodies should not be permitted), ‘non-sensitive’ (where discharge is not a problem due to a considerable dilution of highway pollution) and ‘study’ (where a decision needs to be based on further data collection) .Dynamic surface waters were divided into permanent and ephemeral systems with the former further sub-divided into high and medium flow rivers. Seasonal flow variability is common in most Mediterranean countries and the occurrence of lowest monthly averaged flows above 20 m^3/s categorises a fast flowing river where the combined effect of dilution and hydrodynamics should ensure that the discharge of non-treated road runoff is not a problem. However, balanced against the dilution of soluble pollutants (which occurs during high flows) is the impact on particulate pollutants due to increased resuspension rates and subsequent sediment transport. In the case of medium flow rivers (for which the lowest of the monthly averaged flows are below 20 m^3/s) a specific site analysis is required to access the sensitivity. For ephemeral water bodies further studies are considered to be necessary based on an assessment of the soil infiltration capacity in the riverbed to differentiate between possible impact on the underlying groundwater (high infiltration rate) or the impact on the downstream environment (low infiltration rate).

An existing Slovenian decree has set a limit of 12,000 prescribed passenger cars (AADT equivalent) for the disposal of highway runoff to a flowing stream but a more scientifically based guideline is that 10% of the average low discharge of the receiving stream should not be exceeded by the discharge from the road runoff.

3.1.2 Geological impacts

The physical and chemical composition of surface waters are influenced by the weathering of bedrock minerals, atmospheric processes including evapotranspiration, deposition of dust and salt by wind, the natural leaching of organic matter and nutrients from soil, run-off due to hydrological factors, and biological processes (Khatri and Tyagi, 2015). These natural processes cause changes in the pH, alkalinity and hardness of the water as well as affecting the presence of both dissolved substance and particulate matter. Minerals and dissolved salts are necessary components of good quality water as they help maintain the health and vitality of organisms present. However, some aquatic ecosystems can tolerate large changes in water quality without the composition and function of the ecosystem being affected.

Cations commonly found in surface waters from water–rock interactions include Ca^{2+} and Mg^{2+} ions in limestone/chalk areas, Na^+ ions particularly from plagioclase in granite rocks and K^+ ions from orthoclase and muscovite minerals present in granite. Natural sources of chloride include rainfall, dissolution of fluid inclusions and chloride-bearing minerals with other anions (e.g. CO_3^{2-}, HCO_3^-, SO_4^{2-} and F^-) occurring as a result of run-off from the weathering of appropriate rocks and soils. Dissolved organic matter leached from soils affects the functioning of aquatic ecosystems by influencing acidity and trace metal transport (Evans et al., 2005). Metal toxicity and availability in aquatic systems has been shown by a number of workers to be inversely related to water hardness due to the competition that exists between the metal cations and Ca^{2+} and Mg^{2+} ions which are present as a result of the hardness (e.g. Giesy et al., 1978). Similarly, an increased water acidity promotes the availability of the free metal ions which are widely believed to be the most bioavailable and toxic metal species. In contrast, naturally occurring organic materials , such as humic and fulvic acids, can form complexes and chelates with metals thereby influencing their transport, availability and toxicity. In practice, the combined effects of the various water chemistry parameters need to be assessed to interpret the toxicity of metals to aquatic organisms. Erickson et al. (1996) have investigated the toxicity of copper to larval fathead minnows. Increased pH, hardness, sodium, dissolved organic matter, and suspended solids each caused toxicity to decrease on the basis of total copper concentrations whereas added potassium resulted in increased toxicity. Alkalinity had no observed effect on total copper LC50s, but its effects might have been masked by those of the cations added with it. The results of this study confirmed that although toxicity is clearly influenced by water chemistry the test organism toxicity could not be solely attributed to the cupric ion but to the variety of copper species contributing to the toxicity. The role of water chemistry in influencing the toxicity of metal pollutants to different aquatic species is discussed in more detail in Section 3.1.4.

3.1.3 Hydrological impacts

The effectively impervious surfaces associated with highways have a significant hydrological effect through a reduction in the availability of infiltration and consequently substantial increases in the volumes of surface runoff produced (Fletcher et al., 2013). Different investigations of the infiltration and run-off processes in the UK (Ragab et al., 2003) and France (Ramier et al., 2011) have shown that annual rainfall losses of the order of 30 to 40% occur with typically 20 to 24% lost to evaporation and 6 to 20% lost by infiltration through the road surface. The 60-70% of the total rainfall which constitutes runoff leads to consequences for the receiving waters in terms of ecological impacts (including loss of sensitive species) (see Section 3.1.5), increases in potential toxicants (see Section 3.1.4), and a loss of organic matter (Walsh et al., 2005). An understanding of the mechanisms by which highway runoff impacts receiving waters is essential to assist the development of strategies to protect waterways from further degradation. Wenger et al. (2009) have identified eight aspects which can be associated with the impacts of highway discharges together with the causes leading to them (see Figure 3.1). There are many stressors that can lead to problems for receiving streams and although this report categorises these (e.g. water quality, hydrology, geomorphology, physical habitat), it is important to be aware of the strong overlaps that exist between them.

The hydrological changes resulting from the presence of highway surfaces can lead to significant geomorphic changes, including habitat simplification, scouring of sediments and the creation of larger, deeper channels. The enhanced shear stress generated within the receiving stream due to increases in both the frequency and magnitude of elevated flows results in an increase in the mobilisation and transport of sediments (Bledsloe, 2002). In addition, the existence of the impervious highway surface (and also possibly in the channel itself, as a result of channel lining) results in a reduction in the supply of coarse sediments. Although there may be a temporary increase in the occurrence of coarse sediments during and immediately following construction and/or repair activities (Nelson et al., 2002). The general trend will be a progression towards finer sediments on the channel floor, assisted by the increased concentrations of fine suspended sediments in the highway runoff (Duncan, 1999), in contrast to the starvation of coarse sediments. The increased stream power experienced by receiving waters results in an increased potential for channel migration which may be counteracted by the use of lining (e.g. with rock, concrete or other durable materials).

Such channel modifications, along with the natural loss of channel diversity as a result of reduced coarse mobile sediments lead to substantially degraded habitat quality (Poff and Allan, 1995).

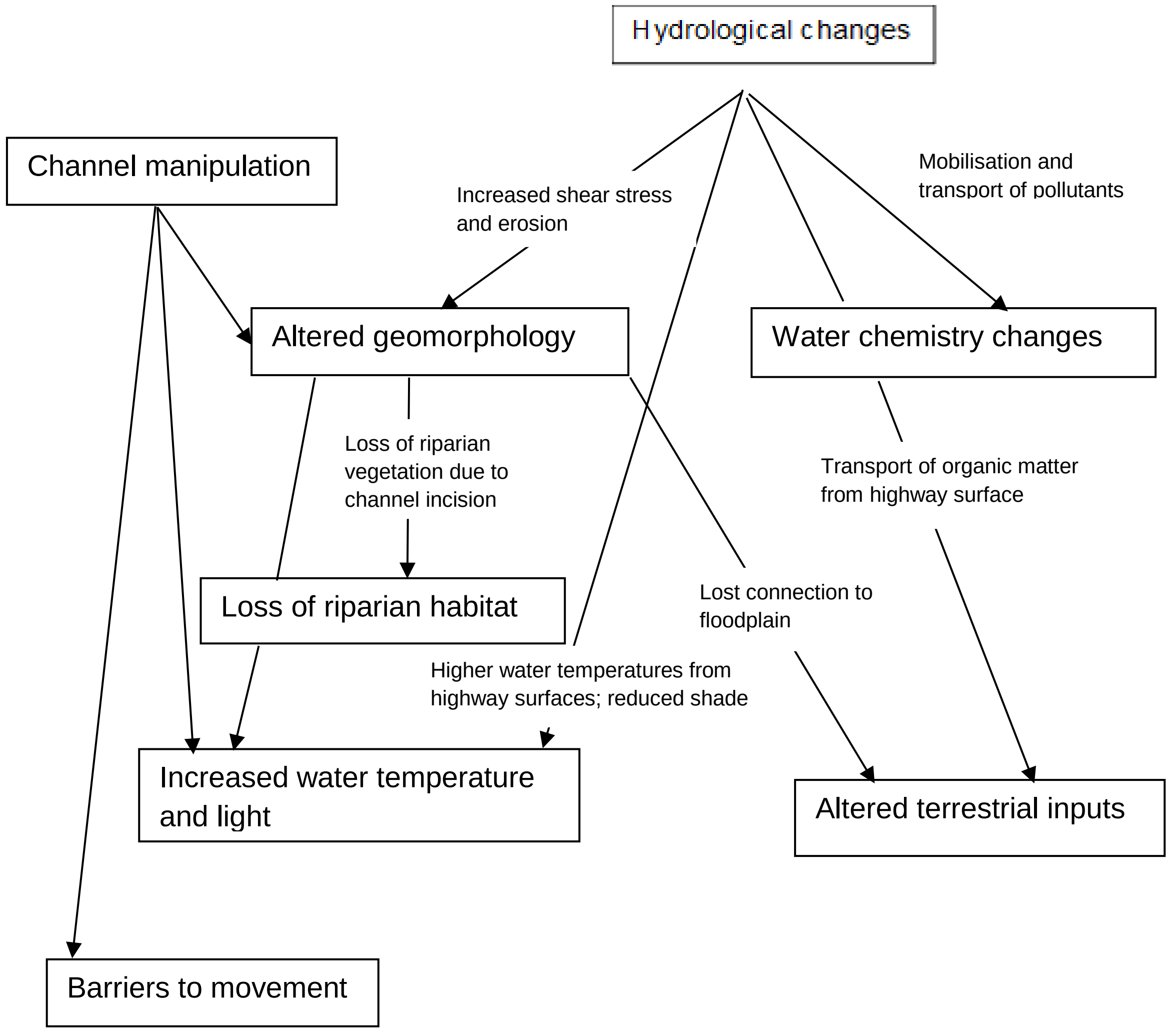

Figure 3.1. Identification of some of the important processes by which the presence of highway surfaces can produce changes in receiving water systems (after Fletcher et al., 2013).

The construction of highways can lead to streams being separated from floodplains, with impacts not only for exchanges between the channel and riparian zones, but potential loss of important ecosystem services delivered by the riparian zone, such as the facilitation of denitrification, particularly during low flows (Groffman and Crawford, 2003).

A potential advantage of discharging highway runoff into surface receiving waters, particularly rivers and streams, is that the resulting dilution may result in pollutant levels which are below the toxic thresholds for even the most sensitive biota. This practice is widely used for treated sewage effluents, where because of the relatively uniform discharge rates and reasonably consistent pollutant levels, it is feasible to estimate the required flow ratios for safe disposal. However, this is not the case for highway runoff where the flows vary throughout individual storm events and pollutant levels are dependent on many factors including rainfall intensity and the length of the antecedent dry period during which pollutants have built up on road and associate surfaces (see Deliverable D1.1). Many researchers have reported the existence of a 'first flush' effect (e.g. Stenstrom et al 2001) in which a substantial proportion of the total pollutant load is discharged during the early stages of a storm event. This could have serious short term consequences for the receiving water biota, particularly in the immediate vicinity of the discharge point. However, in these circumstances a remediating effect can be the additional dilution capacity as a result of the higher in-stream flows during rainfall conditions. This will only be maximised when there is coincidence of peak flows for both the highway runoff and the receiving water with the latter being dependent on the time of concentration for the specific catchment. Perdikaki and Mason (1999) observed that the most notable downstream impacts on biota (as expressed by depressed BMWP scores) due to road runoff discharges were during low flow conditions. Low receiving water flow rates can also lead to water temperature increases and subsequent impacts on water quality such as higher concentrations of dissolved substances and decreased levels of dissolved oxygen (Prathumratana et al.. 2008).

First flush events often coincide with the initial storm events of the winter season and are referred to as the seasonal first flush. Lee et al. (2004) analysed four major data sets, collected during the wet seasons between 1999 and 2003. Pollutant concentrations in the first part of the wet season were between 1.2 and 20 times higher than concentrations near the end of the season with mass emission rates being similarly elevated. The seasonal first flush was particularly pronounced for organics, minerals and heavy metals except for Pb. A similar result for predominantly particulate associated Pb has been reported for a different study (Sansalone and Buchberger, 1997) whereas the dissolved fractions of Zn, Cd and Cu exhibited a strong first flush in lateral highway runoff flow. Aryal et al. (2006) analysed six runoff events to investigate the characteristics of particle-associated PAHs in first flush. Although the fine fraction (<45 μm) had a relatively PAH higher contribution than the coarse fraction, it was the latter which demonstrated fluctuations during the runoff events with the higher loading being concomitant with the initial rapidly increasing runoff flows. Lau et al (2002) observed a first flush phenomenon for PAHs (as well as chemical oxygen demand and oil/grease) in the runoff from three highway sites over two wet seasons. The mass first flush ratio (the ratio of the normalized transported mass of pollutant to the normalized runoff volume) generally was above 1.8 for the first 25% of the runoff volume, and in some cases as high as 2.8. Because of the variability in pollutant concentrations and loads, toxicity can also vary considerably throughout storm events. Kayhanian et al. (2008) found that the first few collected samples were 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. The clear association of the toxicity with the first flush identifies the benefits of effectively treating the first portion of the stormwater runoff volume and the use of stormwater Best Management Practices (BMPs) early in the season could remove several times more pollutant mass than randomly timed or uniformly applied BMPs.

3.1.4 Chemical impacts

The water quality in receiving waters receiving highway drainage is subject to degradation through the increased generation of pollutants as a result of highway activities and the increased mobilisation and transport of these pollutants as a result of increased surface runoff and hydraulic efficiency within the system. In the case of metals, their toxicity can be influenced by a range of different water quality parameters which influence the metal speciation. This defines the different physicochemical forms in which a metal can exist and exerts an important role regarding the behaviour of a metal and hence its impact in any environmental system (Batley et al., 2004). The partitioning of metals between these different physicochemical forms (e.g. truly dissolved, complexed, associated with colloids and particles) is a critical factor in the determination of potential metal bioavailability.

The site-specific water quality parameters which particularly influence metal speciation and bioavailability include water hardness, alkalinity, pH, and natural organic matter (Campbell, 1995; Markich et al., 2003; Janssen et al., 2003; De Schamphelaere and Janssen, 2004). In the case of water hardness, quantitative relationships (algorithms) have been established to describe the reduction in the bioavailability of cadmium, chromium(III), copper, nickel, lead, and zinc as a function of increasing water hardness. Hardness is the only water quality parameter which has been incorporated in the establishment of water quality standards for natural surface waters and in some countries hardness algorithms have been adopted to assist in the regulation of metals (e.g. Australia, New Zealand, USA and Canada). The widely held understanding of the role of water hardness on metal toxicity is interpreted in terms of the increasing competition of calcium and magnesium for metal-binding sites at gill or cell membranes. This is the basis for the predictions developed in biotic ligand models (Paquin et al., 2002).

Although water hardness is the only physicochemical variable currently considered for regulating divalent metal exposures in receiving waters, other variables, such as pH, alkalinity, and dissolved organic matter, may also have an important influence on metal speciation and bioavailability in solution. There is some concern that determinations of toxicity of metals to freshwater biota may have confounded the effects of true water hardness (calcium and/or magnesium concentration) with alkalinity (carbonate concentration) and pH (proton concentration). This is because an increase in calcium and/or magnesium concentration is frequently associated with an increase in alkalinity (as calcium and/or magnesium carbonate) and, hence, pH. Therefore, it is important to separate the effects of water hardness and alkalinity as each variable has a different mechanism of reducing metal toxicity. Although the effect of calcium and magnesium is accepted as competitively inhibiting metal uptake and, hence, toxicity (Varengo et al., 1996), alkalinity directly affects metal speciation in solution through complexation with carbonate, which impacts on metal bioavailability (De Schamphelaere and Janssen, 2002). These mechanisms need to be uncoupled if their effects are to be incorporated in models that can be applied to metal regulation (such as biotic ligand models), which use physicochemical variables to predict the acute toxicity of metals, such as copper, to freshwater biota on a site-specific basis. There is now evidence that in some case, e.g. the toxicity of copper to the cladoceran *Ceriodaphnia dubia*, increasing water hardness does not significantly alter the copper toxicity in synthetic waters (Naddy et al., 2003).

Water pH is an important factor affecting the toxicity of metals to freshwater biota, although a relationship between the two has been difficult to establish. Metals are generally more toxic at low pH due to an increased predominance of the more bioavailable free-metal ion (Morel, 1983). However, it has also been shown that the toxicity of metals to cladocerans can decrease with decreasing pH (Schubauer-Berigan, 1993; Janssen et al., 2003), as a consequence of reduced metal uptake due to competition with the H^+ ion at cell surfaces. Changes in pH for waters close to neutral may influence metal speciation through effects on hydrolysis and ligand binding. Although not as influential as water hardness and pH, total suspended solids (TSS) can have a protective role with respect to metal toxicity to fish (Erickson et al., 1996). However there is a threshold effect after which physical irritation to fish gills counteracts the benefits imposed by TSS. Natural dissolved organic carbon (DOC), in the form of fulvic and humic acids, is an important complexing agent for some metals, such as copper, in aquatic systems (Tipping, 2002). Metal complexation by DOC has been shown to reduce metal toxicity in freshwater biota (Kim et al, 1999), via reduction in the free-metal concentration. Although numerous studies have reported the effects of natural DOC on metal toxicity to freshwater biota, the vast majority has not determined metal speciation, and thus they are qualitative in nature. Consequently, they are not suitable for testing biotic ligand models.

Hyne et al. (2005) have investigated the influence of pH, alkalinity, dissolved organic carbon (DOC) and hardness on the aqueous speciation of copper and zinc and their relationship to the acute toxicity (48 hour immobilisation tests) to the cladoceran *Ceriodaphnia cf dubia*. The toxicity of copper decreased fivefold with increasing pH, whereas the toxicity of zinc increased fivefold with increasing pH (due to a decrease in competition between H^+ ions and Zn^{2+} ions for the receptor binding sites). Increasing DOC (by adding natural fulvic acid to synthetic water) was found to decrease linearly the toxicity of copper, whereas the addition of DOC only resulted in a very small reduction in the toxicity of zinc to *C. cf dubia*. Copper toxicity to *C. dubia* generally did not vary as a function of hardness, whereas zinc toxicity was reduced by a factor of only two, with an increase in water hardness from 44 to 374 mg $CaCO_3$/L. Increasing bicarbonate alkalinity of synthetic waters (30-125 mg/L as $CaCO_3$) decreased the toxicity of copper up to fivefold, which was attributed to the formation of copper-carbonate complexes, in addition to a pH effect. The toxicity of copper added to a range of natural waters with varying DOC content, pH, and hardness was consistent with the toxicity predicted using the data obtained from synthetic waters.

In fish and other aquatic organisms, the acute lethality of metals, such as copper, is known to be mediated by the gill epithelium. The influence of water quality parameters (including water hardness, alkalinity, pH, and dissolved organic carbon) is commonly estimated using biotic ligand models (BLM) which are able to predict the susceptibility of the toxicity to varying chemical compositions (Santore et al., 2001; Niyogi and Wood, 2004). Cations (e.g., Ca^{2+}, Mg^{2+}, and Na^+) reduce the bioavailability of metal ions to the binding site (biotic ligand) on the gill by competition, whereas anions (e.g., HCO_3^{2-}, CO_3^{2-}, Cl^- and SO_4^{2-}) and dissolved organic carbon (DOC) bind to the free metal ions to form inorganic and organic complexes which also reduce the metal availability to the biotic ligand thereby reducing the lethal effect of metals. Quantitative DOC studies are not always consistent with biotic ligand model predictions and therefore further evidence is required to test the validity of these models with regard to the role of natural DOC in aquatic systems. Biotic ligand models employ stability constants representing the strength of binding of cations to the ‘biotic ligand’ to predict the relationship between water hardness and toxicity of metals (Di Toro et al., 2001). For Ni, a priority substance under the EU Water Framework Directive, bioavailability models have been developed for fish (Deleebeeck et al, 2007), crustaceans (Deleebeeck et al, 2008) and algae (Deleebeeck et al, 2009).

In northern climate countries there have been large increases in the use of road de-icing salt during winter months over the last 60 years in order to keep roads safe. Kausal et al. (2005) have demonstrated that chloride levels in rivers and streams correlate with the percentage of impermeable surfaces in a catchment. Increases in freshwater salinity has implications for ecosystem health such that streams adjacent to highways frequently exceed the levels considered harmful to aquatic life (Trowbridge et al, 2010). The occurrence of elevated chloride levels in small, deep water bodies can also inhibit the seasonal turnover of lake waters, with basal saline waters becoming oxygen depleted additionally leading to the release of nutrients by sediments over time. However, the impact of elevated salt levels on receiving water biota appears to vary significantly according to the species affected. Hurle et al., (2006) demonstrated that fish were able to tolerate short-term increases in sodium chloride concentrations whereas algae and some macroinvertebrates were considerably more sensitive. Comparison of two algal species, *Synedra delicatissima* and *Selenastrum capricornutum*, showed that the former was considerably more sensitive to sodium chloride indicating that runoff induced by heavy rainfall immediately following salt application to a highway could exert a serious impact on this algal species. Macroinvertebrates have been shown to exhibit drift behaviour in streams receiving runoff from major highways due to high chloride levels (Blasius and Merritt, 2002). Toxicity tests have shown that both *Daphnia magna* and *C. dubia* are affected by elevated concentrations of salts in stormwater runoff (Novotny and Witte, 1997) with the chronic test for the latter being more sensitive than the *D. magna* acute test when evaluating toxicity of runoff resulting from road salts (Mayer et al., 2011). Gillis (2011) have identified that chloride can have a negative impact on freshwater mussels with acute toxicity occurring when chloride levels reach 1300 mg/L.

In addition, increases in the base concentration of chloride in streams near roads that receive de-icing salts has implications for the fluvial transport and partitioning of metals between the dissolved and suspended phases. The formation of chloride complexes facilitates the transfer of metals which are initially preferentially adsorbed to suspended particulates to the dissolved phase and enhances their potential bioavailability to aquatic organisms (Begeal, 2008). Morrison et al (1986) have monitored similar processes in stormwater runoff with increases in different metal species being observed as chloride concentrations approached levels of 10,000 mg/L during rainfall events accompanying snowmelt after winter applications of de-icing salt. The resulting high ionic strength assisted the desorption of metals attached to suspended solids into the soluble phase especially for Cd and Pb due to the formation of strong chloride complexes. This process was less pronounced for Cu and Zn due to the counteractive impact of particulate organic carbon attached to the solid phases. A study of the pathways of pollutants in urban snow deposits found that while the majority of metals in the snow were attached to particles more than 50% were in the dissolved form in the melt water (Viklander, 1996). Also in Sweden, Bäckström et al. (2004) have investigated the seasonal variations of Cd, Cu, Pb and Zn in soil solutions as a function of distance from roads. The metal concentrations increased during the winter, but through different mechanisms. Cadmium concentrations in the aqueous phase increased as result of ion exchange processes accompanied by the formation of chloride complexes. Similarly, the concentrations of zinc increased, due to ion exchange, with calcium and protons. The mechanisms of mobilisation for copper and lead were less clear probably due to association with coagulated or sorbed organic matter in combination with colloid dispersion.

Due to their strongly hydrophobic characteristics, polyaromatic hydrocarbons (PAHs) tend to be strongly sorbed to particulate material and are considerably less influenced by any changing water qualities than is the case for metals. The main parameters affecting the distribution of PAHs between the solid and soluble phases is organic carbon. The affinity of PAHs for sediments is enhanced by particulate organic carbon and Northcott and Jones (2001) have shown that the release of PAHs to the dissolved phase decreases with increasing organic carbon normalised soil–water partitioning coefficient (K_{oc}) values as well as with increasing octanol–water partitioning coefficient (K_{ow}) and molecular weight. The presence of dissolved organic carbon in the waters phase is known to be able to increase the mobility of hydrophobic organic compounds, such as PAHs, through binding mechanisms to soluble macromolecules, such as humic and fulvic acids (Naes et al., 1998). Increasing the ionic strength, which would occur due to the presence of salt in the receiving water, can result in additional mobilisation of DOC from the sediments due to chemical interactions with Na ions whereas Ca ions enhance the precipitation of DOC. Using column leaching experiments, Revitt et al. (2014) have shown that the influence of increased mobility of DOC as a result of increased salinity had some impact for the lighter and more soluble PAHs (fluoranthene and pyrene) but was negligible for the heavier PAHs [benzo(e)pyrene and benzo(ghi)perylene].

Hiki and Nakajima (2015) have investigated 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 an amphipod after 10 days of exposure with the high mortality at 35‰ salinity being attributed to aquatic exposure as opposed to dietary exposure. However, although PAHs will have played a role in this toxicity it is likely to be the presence of metals in the contaminated road dust which has the major impact on benthic organisms at high salinity levels. The potential contributing role of PAHs in the toxicity of highway runoff has been demonstrated by Boxall and Maltby (1997) with six PAHs (anthracene, phenanthrene, fluoranthene, pyrene, chrysene and benzo(a)anthracene) being pinpointed of which two (fluoranthene, and pyrene) were the most toxic. This was confirmed by Crabtree et al. (2009) in an analysis of 340 UK highway runoff events although for neither fluoranthene nor pyrene did the measured event mean concentrations exceed the 6 hour Runoff Specific Threshold values (see also Section 3.1.2.1; Deliverable D2.1).

3.1.5 Ecological impacts

The ecological impacts of highways can be extremely varied and include effects on roadside vegetation and animals as well as population influences through, for example, road kills, road avoidance due to traffic disturbance and habitat fragmentation due to barrier effects (Forman and Alexander, 1998). Trombulak and Frissell (2000) have also reported that roads promote the dispersal of exotic species by altering habitats, stressing native species, and providing movement corridors. As a consequence the presence of roads is strongly correlated with changes in species composition, population sizes, and the hydrologic and geomorphic process that shape aquatic systems. However, of specific interest to this report are those ecological impacts relating to the receiving water environment and the influences on sediments and chemical transport due to discharges from highway surfaces. Consequently, there will be overlaps with the material covered in both the Hydrological Impacts (Section 3.1.3) and Chemical Impacts (Section 3.1.4) sections but it is the intention here to focus specifically on how these affect the receiving water ecology. Stream ecosystems subject to highway discharges generally have altered flow rates, pool riffle sequences and scour, which contribute to a reduction in habitat and aquatic organisms. Increased sediment transfer disrupts stream ecosystems by inhibiting aquatic plants, macroinvertebrates and fish. During low flow periods, fine sediment deposits tend to fill pools and smooth gravel beds leading to degradation of habitats and spawning sites for fish species. Salmonids and other riverine fish actively move into seasonal floodplain wetlands and small valley-floor tributaries to escape the stresses of main-channel flood flows but valley-bottom roads can destroy or block access to these seasonally important habitats (Copp,1989). Persistent barriers may encourage local selection for behaviours that do not include natural migration barriers, potentially reducing both the distribution and productivity of a population.

Aquatic invertebrates represent a diverse group of taxa that live in, on, or near streambed sediments and include algae, benthic invertebrates, and fish. The different taxonomic groups respond differently to natural or anthropogenic disturbances because of differences in habitat, food, mobility, physiology, and life history. 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. Some genera are highly sensitive to changes in the aquatic environment 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 are only 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 (see Section 3.1.2.3; Deliverable 2.1). It can be difficult to isolate the impacts due to highway runoff from other factors. Thus, Bruen et al. (2006) found that none of three investigated fish species demonstrated a negative impact due to highway runoff although this may have been masked 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 ten different macroinvertebrate fauna with identified differences being assigned to limitations in the physical habitat or to impacts from other sources.

The availability of relevant food sources can influence the distribution of different aquatic genera. For example, macroinvertebrate diversity is known to be related to their ability to process leaf litter with reductions occurring as a result of changes from the presence of benthic algae and coarse particulate organic matter to one dependent upon fine particulate organic matter and dominated by collectors as opposed to shredders (Magurran, 1988). 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 due to alterations in water quality (Englert et al., 2015). 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 exert important ecological impacts on aquatic invertebrate communities impacted by highway runoff although this may be partly mediated by increased microbial activity (leading to the higher leaf decomposition) at the downstream site (Maltby et al., 1995).

Ecological impacts may also depend on the life cycle stage of an individual species. Studies of pollutant uptake by the frog have shown that the steepest increases were observed between the developmental stages of egg and tadpole reflecting fundamental physiological differences between these two developmental stages (Meland et al., 2013). 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 the gills and passive diffusion through the body surface.

3.1.6 Summary of receiving surface water impacts and vulnerabilities

In Sections 3.1.2 to 3.1.5 the roles of geological, hydrological, chemical and ecological impacts in influencing the status of surface receiving waters have been discussed. The findings are summarised in Table 3.1, which also explains whether each of the identified specific processes exerts either a beneficial or detrimental impact as far as the receiving water is concerned. Finally, a score is allocated to each impact on a scale of 0 to 5 with an increasingly +ve score indicating increasing improvements in the receiving water and an increasingly –ve score signifying greater potential problems for the survival of biota in the receiving water.

Factors which potentially reduce the risk posed by highway runoff discharges to receiving streams include the possibility of an increased dilution capacity (+3), the presence of elevated hardness in the receiving water (+2) and/or alkalinity (+1) and the provision of essential minerals and nutrients due to the natural weathering of rocks and soils (+1). In contrast, there are considerably more factors which can result in an increased risk being posed to the receiving water environment as a consequence of highway runoff discharges. Of particular consequence is the presence of elevated concentrations of chloride in highway runoff due to winter maintenance activities (-4) as this can directly affect aquatic biota resulting in drift behaviour as well as facilitating increased toxicity through the release of particulate associated metals and PAHs to the soluble phase. The existence of a first flush effect in which elevated loads of toxic pollutants are discharged to a receiving water during the early stages of a storm event also presents a high risk scenario (-3). Detrimental risk scores of -2 have been awarded to impacts related to increased flows which can result in both changes to the receiving water habitats associated with pool riffle sequences and the increased mobilisation and transport of sediments. Changes in the ecological status of receiving water flora and fauna due to physiological stresses induced by factors such as increases in water temperature and water quality variations are also considered to present a risk score of -2. Elevation of receiving water acidity could represent a similar risk score but the toxic impact of the subsequent release of free metal ions can be ameliorated to some extent by an increased competition for cell surface adsorption sites by hydrogen ions thereby meriting a risk score of -1. Similar risk scores are awarded to the ecological impacts created by the physical presence of roads (encouraging increases in non-native species and potential losses of ecosystem services) and reductions in biodiversity due to changes in food availability.

Table 3.1. Summary of the impacts and associated vulnerabilities arising from the discharge of highway runoff to receiving waters

Type of impact	Influencing factor	Nature of receiving water impact	Beneficial or non-beneficial impact	Score (+ = beneficial; - = non-beneficial)
Geological (see Section 3.1.2)	Natural weathering of rocks and soils	Provides essential minerals and nutrients	Contributes to maintaining the health and vitality of aquatic ecosystems	+1
Hydrological (see Section 3.1.3) due to highway runoff inputs	Increased flows	Changes to pool riffle sequences	Reduction in receiving water habitats	-2
	Scouring of basal sediments	Increased mobilisation and transport of sediments with a shift from coarse to fine sediment distribution	Loss of channel diversity due to channel migration), substantially degraded habitat quality,; inhibition of aquatic plant, macroinvertebrate and fish growth (reduction in spawning sites)	(-2)
	Enhanced dilution capacity	Potential to reduce pollutant levels below toxic thresholds	Protection of sensitive biota	+3
	First flush effect	Possibility of high pollutant loads being delivered to receiving water during early stages of a storm event (particularly after long dry periods)	Elevated levels of toxicity during initial stages of a storm event	-3
Chemical (see Section 3.1.4)	Increased total suspended solids (TSS) levels in highway runoff	Particulates (particularly those enriched in organic material) acts as sites for both metal and PAH sorption	Possible small reduction in toxicity due to pollutant adsorption likely to be balanced by increased turbidity (habitat degradation) and biotic impacts such as physical irritation to fish gills	0
	Elevated sodium chloride (salt) concentrations in highway runoff due to winter maintenance activities	a) Aquatic biota demonstrate different sensitivities to high salt levels b) Chloride can facilitate the release of metals from solid to soluble phase due to ion exchange processes and formation of chloro-complexes c) Increased ionic strength can mobilise dissolved organic carbon (DOC) from solid phase and associated release of PAH to soluble phase	Encouragement of drift behaviour and possibility of toxic impact Increased toxicity Increased toxicity	-4

	Receiving stream water hardness (typified by presence of Ca and Mg ions)	Ca and Mg ions compete with metal pollutants for membrane binding sites in aquatic organisms	Reduction in metal bioavailability and toxicity	+2
	Receiving stream alkalinity (typified by carbonate concentrations)	Removal of free metal ions due to carbonate complexation	Reduction in metal bioavailability and toxicity	+1
	Presence of naturally occurring dissolved organic materials (e.g. humic/fulvic acids) in receiving water	a) Removal of free metal ions due to organic complexation b) Mobilisation of hydrophobic PAHs previously preferentially attached to particulates	Reduction in metal bioavailability and toxicity Potential increases in PAH bioavailability and toxicity	0
	Receiving stream acidity (typified by hydrogen ion concentrations)	Promotes availability of free metal ions; some competition between metal and hydrogen ions for cell surface adsorption sites	Increase in metal toxicity balanced by some competitive uptake.	-1
Ecological (see Section 3.1.5)	Elevated temperatures in highway runoff	Influences physiological processes; reduces dissolved oxygen levels and possibly increases concentrations of dissolved substances	Affects the metabolic and reproductive rates of algae, benthic invertebrates and fish	-2
	Physiological stresses induced by changing water quality	Species sensitivity	Different species and different life stages show different responses	-2
	Physical presence of roads	a) Provide movement corridors and changes to stream habitats b) Separation of streams from flood plains	Encourage changes in species composition involving removal of native species and invasion by non-native species Limitation of exchanges between stream channel and riparian zone leading to potential loss of ecosystem services	-1
	Factors (e.g. coarse/fine sediment deposition) which influence food availability	Affects the distribution of different aquatic species	Possible reductions in biodiversity	-1

It is evident from the overall consideration of risk scores that there are identified impacts of roads and road discharges on receiving waters which can result in conflicting detrimental and beneficial impacts. This is particularly true of changing flow conditions (both volume and turbulence effects) (Table 3.2) occurring as the discharge enters the receiving stream. The enhanced dilution capacity can be beneficial in terms of reducing the pollutant concentrations below toxic levels whereas the occurrence of turbulent conditions can lead to scouring of basal sediments and changes to pool riffle sequences both of which pose greater risks to habitat preservation. Therefore high intensity discharges can potentially cancel out the advantages provided by dilution and ideally should be avoided by, for example, the introduction of an appropriate storage system between the runoff outlet and the receiving water. Such systems also provide the added benefit of water quality improvements.

Table 3.2. Overview of potential conflicting impacts with regard to the risk posed to receiving waters by highway runoff discharges

Influencing factors	Identified impact	Allocated risk score
Increased flows	Dilution	+3
	Scouring	-2
	Changes to pool riffle zones	-2
Pollutants in highway runoff	Elevated chloride levels	-4
	First flush effect	-3
	Induced receiving water quality changes	-2
	Raised water temperature	-2
	Increased SS levels	0
	Increased acidity	0
Characteristics of receiving water	Elevated hardness	+2
	Elevated alkalinity	+1
	Presence of natural DOM	0

As expected, the greatest risks to the receiving water environment occur as a consequence of elevated pollution levels in the highway runoff (Table 3.2). Elevated chloride levels, as a result of winter maintenance activities, pose a serious risk to receiving water biota in terms of an increased toxicity through at least 3 different pathways (see Table 3.1). Because of its high solubility, the impacts of chloride are difficult to control by treatment of the runoff and the best option is to instigate appropriate methods of source control (e.g. limiting the use of chloride based de-icing agents or employing different de-ice-ants). The presence of a first flush effect can represent a severe risk to the receiving water during the early stages of a storm event due to both elevated flows and the delivery of high pollutant loads during a short time period. The seriousness of this impact can be controlled to some extent by providing appropriate storage of the highway runoff particularly during the early stages of storm events. The ability to provide storage of highway runoff prior to discharge also has the potential to ameliorate the physiological risk arising from increased temperatures in the receiving water.

The existing chemical characteristics of the receiving water generally have the ability to reduce the toxic risks posed by elevated metal concentrations in highway runoff (Table 3.2). The increased presence of Ca and Mg ions associated with higher water hardness levels compete with metal pollutants for membrane binding sites in aquatic organisms. Similarly, higher alkalinity reduces metal bioavailability and toxicity due to possible carbonate complexation of free metal ions introduced into the receiving water. The presence of naturally occurring dissolved organic materials can similarly produce a reduction in metal toxicity but this is balanced by a potentially increased risk due to the release of PAHs adsorbed to particulates.

3.2 Groundwater

3.2.1 Introduction

The EU Groundwater Directive; 2006/118/EC (2006) essentially charges national regulatory authorities to take a risk-based approach to "prevent and limit" the entry of hazardous substances into groundwater. However, given the size, scale and, perhaps above all, the relatively slow mixing of resident water within groundwater bodies, it is quite possible for site-specific contaminant discharges such as associated with "point source" highway drainage, to result in localised pollution whilst the areal/regional groundwater body maintains a good chemical status. It is this localised focus which drives the core interest in and concern for highway drainage and particularly the potential for localised groundwater pollution from infiltration-type mitigating control systems. The popularity of such drainage systems is evidenced for example, by the occurrence of over 60 soakaways in a 31.6 km stretch of the M25 London Orbital Motorway between Junction 18 – Jct.24; all being located over a designated highly sensitive groundwater protection zone (SPZ) in the underlying Chalk aquifer.

In recent years, the use of sustainable drainage systems (SuDS) which involve use of appropriate treatment systems in series prior to disposal has become widespread. These control and treat highways runoff prior to discharge to a waterbody. If such systems do present a long term source of groundwater pollution, their ubiquitous and dispersed distribution along the national European highway network might become difficult and costly to retrospectively address and rectify. This accentuates the need to understand and monitor the driving processes of subsurface attenuation and degradation within the critical unsaturated zone beneath such highway drainage infiltration systems. This groundwater section examines the basis for the identification and characterisation of the critical controlling factors influencing the impact of highway-derived drainage contaminants on the quality of underlying groundwater bodies. It is not concerned with the detailed biogeochemical properties and dynamics or modelling protocols of the processes themselves, but with the delineation of their defining generic characteristics and significance for risk assessment approaches.

3.2.2 Modelling approaches for the identification and application of critical factors influencing groundwater pollution potential

Groundwater pollution risk assessment has well developed conceptual and theoretical modelling application methodologies as a basis for regulation, remediation and management of aquifer water quality (McMahon et al., 2001a; Anderson et al., 2015). A wide range of computer modelling techniques and associated spreadsheet approaches are available which provide the basis for the identification and characterisation of critical controlling factors which influence groundwater contamination potential resulting from polluted surface flows such as might occur from highway drainage. The most frequently used impact modelling applications and vulnerability mapping approaches include DRASTIC (Aller et al., 1987), EPIK (Doerfliger and Zwahler, 1997), the AVI method (Van Stempvoort et al., 1993), the Irish method (Daly and Drew, 1999) the PI index (Goldscheider et al., 2000) and others such as GOD (Foster, 1987), COP (Daly et al., 2002), the German method (Holting et al., 1995) etc.. A number of these methodological techniques have been specifically developed to address the particular conditions leading to groundwater pollution risks in areas of fractured/fissured carbonate rocks and karstic conditions. However, intrinsic approaches – such as DRASTIC – are not sensitive to the use of alternative land-use types. Intrinsic vulnerability mapping has been widely used to identify and characterise source protection zones (SPZs) based on relative groundwater sensitivity to contaminant surface releases e.g. in UK (Carey et al., 2017), Ireland (Daly and

Warren, 1998), Portugal (Leitao et al., 2010), the US (USGS, 1999), Canada (Van Stempvoort et al., 1993), Germany (Goldscheider, 2005), France (Plagnes et al., 2010) etc.

Many individual guideline protocols for the assessment of groundwater vulnerability and for mapping sensitive zones have been published by national regulatory authorities including those developed in the UK (Carey et al., 2017), the US (Focazio et al., 2013), Australia (Piscopo, 2001), Portugal and Slovenia (Brencic et al, 2012) as well as Greece (Nanou and Zagana, 2018) amongst many others. Various critiques and reviews have been published covering vulnerability mapping methodologies including more recently those by Maria (2018), Katyal et al., (2017) and Sharazi et al (2012). Such vulnerability mapping is appropriate to the regional level scale (1:50,000 – 1:100,000) and is acceptable for preliminary and general planning purposes. For larger scales (>1:25,000) much more local and site information will normally be required to provide a reliable and robust risk assessment. Such detailed assessment would usually require a process-based modelling approach, particularly for locations which have sensitive groundwater bodies and which may already be subject to pollution impact.

Groundwater vulnerability (GWV) and source protection zone (SPZ) mapping provides the fundamental basis for statutory aquifer protection in many countries and is essentially based on contaminant travel time to the abstraction point. Such protection zones have been defined independently of road infrastructure which for many highways and motorways would have been introduced or been in place prior to their construction. Such legal status applies on a regional scale and requires that site-based highway drainage must comply with statutory water quality standards set for such groundwater zonal protection. The range of protection options range from fixed radius approaches to well-head protection area methods. The SPZs are derived using numerical models of the well flow field in conjunction with particle tracking techniques to delineate the travel endpoints. Most of the approaches incorporate additional “buffer zones” around the well protection areas to allow for methodological uncertainty associated with the modelling parameterisation. There is often a working assumption that the stresses influencing the size and shape of the well capture zone are constant in time and that the aquifer properties influencing the capture zone are spatially homogeneous within the capture zones, are isotropic and have a definable deterministic value. This leads to a specific, but highly uncertain, deterministic delineation of the well capture zone.

The highest level of uncertainty associated with the well capture zone delineation relates to the representation of the hydraulic conductivity distribution and this is particularly the case for Chalk and karst aquifers being highly sensitive to recharge events or to variability in flow direction. In these circumstances, a deterministic approach can be unrealistic. The assumption of a steady state flow field dominating within the capture zone may well be unreliable. It might be better to stipulate the application of transient, integrated surface water/groundwater flow modelling to describe the flow condition which would normally result in a significantly different recharge and groundwater flux estimate in comparison to the traditional methods. Combined with particle tracking analysis, such modelling could well result in more reliable estimates of the recharge area and capture zone. The addition of water chemistry data could then lead to even more accurate and robust vulnerability mapping.

The critical processes influencing the transport and fate of contaminants in groundwater have also been addressed by numerous analytical and numerical modelling tools including the RBCA standard procedure (ASTM, 2000), FLOWPATH II, MODFLOW (McDonald and Harbaugh, 2003) and ConSim v2.5 (Golder Associates, 2003). MODFLOW has been combined with various other contaminant transport codes such as MT3D and MT3DMS to predict the impact of point source pollution. These modelling codes normally incorporate particle tracking techniques to determine groundwater pathway flows and are capable of 2D/3D visualisation as well as being time variant or steady state simulations. Numerous other numerical codes have been developed such as SUTRA, SWIFT, NAMMU, AQUA, AREMOS, SINTACS, PaPRIKa etc., to meet particular circumstances and site-specific conditions. These complex contaminant transport and receptor distribution protocols are intended for site specific, higher tier risk assessment and to help account for properties and processes such as fissure flow, variable density, dual phase flow and unsaturated flows as well as biochemical transformations and generally incorporate probabilistic Monte Carlo uncertainty analysis. It can be argued that methods which assess vulnerability outside of a probabilistic framework are of limited use and can only at best provide a general nonparametric analysis and a limited screening profile. On the other hand, it might be argued that the identification of a single absolute index of vulnerability is defensible as long as preferential flowpaths due to fractures are not present and that low vulnerability refers to susceptibility only to long term conservative pollutants in contrast to high vulnerability implying sensitivity to rapid impact scenarios. Such simplistic index approaches however, are always only likely to be appropriate for preliminary screening and planning purposes.

The definition of groundwater source protection zones (SPZs) has traditionally relied on porous-media based numerical methods to determine zones of advective travel and total pollutant capture. Such models assume that fractured/fissured subsurface rock can be represented by an equivalent porous medium (EPM) at the scale of the numerical model grid cell size. This approach does allow aquifer parameters to be varied to obtain the deterministic uncertainty zones—the zones of confidence, best estimate and zone of uncertainty. A confidence index can then be attached to the overall modelled zones, specified as the ratio of the area of the zone of confidence to the area of the zone of uncertainty. However, the individual deterministic uncertainty zones do not have any confidence attached to them. Robinson and Barker (2000) proposed using a stochastic probabilistic framework to produce a risk based assessment. The methodology requires knowledge of the fracture density and spacing, fracture orientation and length, but is only applicable on a site scale. The fractured nature of carbonate rocks and karst promotes rapid and concentrated movement of recharge waters through the unsaturated zone to the water table minimising opportunities to attenuate surface derived pollutants. In addition, the relatively impermeable matrix of carbonate rocks results in little water-rock interaction. The water quality effects of surface derived contaminant releases can therefore result in short acute episodic impacts through to protracted “diluted” events lasting several days or even weeks. This stochastic probabilistic approach to risk assessment for highway discharges has been demonstrated by studies of field soakaways overlying the UK Chalk aquifer on the M25 London Orbital Motorway in S E England (Robinson et al., 1999).

Many studies have suggested that the probability of determining a specific pollutant compound in groundwater is essentially a function of site adsorption and biodegradation process parameters which are frequently unknown or difficult to quantify and calibrate as well as being time variant (Scott Wilson Ltd., 2009). Solute “piston-flow” transport in the critical unsaturated zone is primarily influenced by advective downward water flux, sorption, retardation and biodegradation reactions and decay processes (Carey et al., 2000; Scott Wilson Ltd., 2009). All these processes are complex, interactive and time/space variant which renders their characterisation to substantial uncertainty. Recent studies have suggested that a front-end conceptual modelling approach prior to the application of a numerical model such as MODFLOW provides a more reliable and accurate simulator of factors influencing groundwater flow. This modelling integration is said to offer a better understanding of site conditions with the ability to select different grid types to reduce model uncertainty whilst facilitating comparison of the simulated results with observed field data.

3.2.3 Approaches to groundwater risk assessment

Under the EU Groundwater Directive (80/69/EEC), risk assessment must establish the acceptability of any new or on-going land use activity to ensure that List I substances do not discharge to groundwater and that any List II substances do not prejudice or lead to poor groundwater quality. Given that the risk assessment should be matched against the likelihood and severity of any pollution impact, most national authorities have adopted a tiered approach (screening, generic and quantitative risk assessment) to the decision-making process. Generic assessment criteria are derived using generic assumptions about the characteristics and behaviour of sources, pathways and receptors. It is assumed in the risk assessment modelling approach that these assumptions will be conservative and falling within a pre-defined range of conditions and values. For higher tier, site-specific assessment criteria, parameter values are identified for individual pollutant concentrations derived from field or other validated data sources and which can be used to quantify unacceptable levels of risk at the receptor point. At each tier the assessment methodology identifies potential hazards and the probable consequences and magnitude of factorial impact to evaluate the significance of any risk. Most also employ a precautionary "allowance" element where it is considered there may be a considerable uncertainty associated with the input data, parameterisation or predicted outcomes.

Table 3.3 illustrates the structure and basis of this tiered hierarchical approach which enables the detailed analysis of the differing factors affecting each component of the risk assessment procedure within the Source-Pathway-Receptor (SPR) system. The table illustrates four stages to the tiered approach as adopted in the UK guidance but some risk assessment methodologies employ several additional levels. The table also lists the basis for the Remedial Targets Methodology (RTM) for phased groundwater risk assessment developed for application by the UK Environment Agency (Carey et al., 2006) and the ConSim modelling approach (Golder Associates, 2003). Figure 3.2 is an outline schematic for a typical groundwater pollution transport model which indicates that the highest priority for protection of the abstraction zone is the primary pollution source around the "entry window" to the subsurface.

Table 3.3 Tiered approach to groundwater risk assessment

Tier	Approach	Focus of approach	Remedial targets method	ConSim modelling approach
Level 1	Screening "areal" approach with "look-up" tables for factor parameterisation.	Pollution source	Target pollutant concentrations in aquifer pore-water (regulatory MAC values)	Comparison of pollution source with receptor
Level 2	Generic quantitative approach to parameter characterisation; site-specific target levels; simple transport modelling and statistical analysis	Flow transport pathway	Target concentrations x Dilution factor (DF)	Unsaturated zone travel time; transport processes; biodegradation; dilution effects
Level 3	Complex quantitative analysis of pollutant fate and transport modelling for site and condition specific circumstances	Receptor Abstraction point	Target concentrations x DF x Attenuation Factor (AF)	Saturated zone transport, attenuation and retardation processes
Level 4			Complex theoretical modelling including spatial variability and probabilistic distributions and uncertainty analysis	Including biodegradation processes

This is particularly the case where there is a likelihood of high magnitude events, such as highway spillages, occurring over a narrow "cone of entry" into the subsurface with acute pollutant release concentrated into a short time period. The groundwater vulnerability will be substantially increased where such rapid acute releases occur over vadose or aquifer zones having fissured soil/rock media.

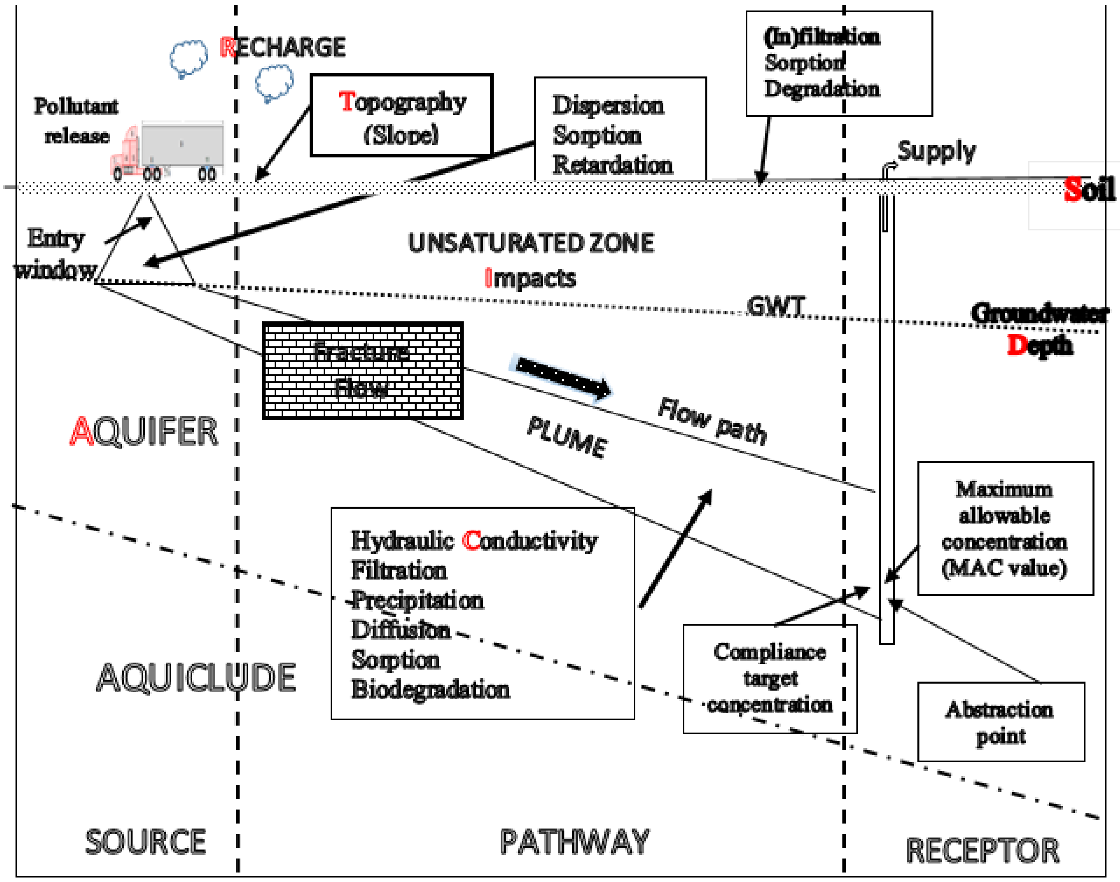

Figure 3.2 Schematic diagram for a groundwater pollution transport modelling approach

Given the possibility of such accidental pollution events, the "entry window" source site becomes the most logical location to implement mitigating controls and systematic maintenance operations, rather than relying on subsurface dilution, attenuation and degradation processes or on costly receptor treatment. The upper case red letters in Figure 3.2 indicate the seven factors that comprise the acronymic basis for the DRASTIC groundwater vulnerability mapping methodology. The majority of such impact risk assessment approaches apply individual parameter weightings and rating coefficients to express relationships between the factors and to reflect their relative importance for groundwater vulnerability assessment. The overall pollution potential or DRASTIC index of vulnerability (IV) is determined for example, by applying an additive formula for the seven factors (see Figure 3.2 and Table 3.5 for symbol key) together with their weightings (W) and ratings (R):

$$IV = (D_R * D_{W}) + (R_R * R_W) + (A_R * A_W) + (S_R * S_W) + (T_R * T_W) + (I_R * I_W) + (C_R * C_W)$$

The weighting and ratings might be varied both spatially and temporally and should not be considered as being rigid unchangeable values. They do not necessarily reduce the parameter autocorrelation or the inherent uncertainty associated with the predicted pollution potential index outcomes which should be viewed with some caution. Some guidance on the allocation of parameter scores and uncertainty values can be obtained in McMahon et al., (2001b). Sensitivity analysis can help the selection of more appropriate parameter values but given the fundamental issue of autocorrelation, care needs to be exercised not to select an "all conservative" suite e.g. a 0.01 hydraulic gradient with a conductivity value of 5mm/day which cannot reflect a real scenario. Figure 3.3 illustrates the variability in risk assessment scores and uncertainty associated with many modelling outcomes for receptor points. Irrespective of the uncertainty level, a high pollution risk assessment score (PRA) would argue for the introduction of acceptable mitigating controls (Sites E and F). Likewise consistently low PRA scores, in the absence of other limiting criteria, would support the exclusion of such controls (Sites A and B). In some instances however, additional evidence might be sought to clarify any uncertainty associated with both high or low PRA scores (Sites C and D). It is fairly easy however, from such a display to identify the sites possessing the highest risk as well as identifying which sites might benefit most from additional data and information.

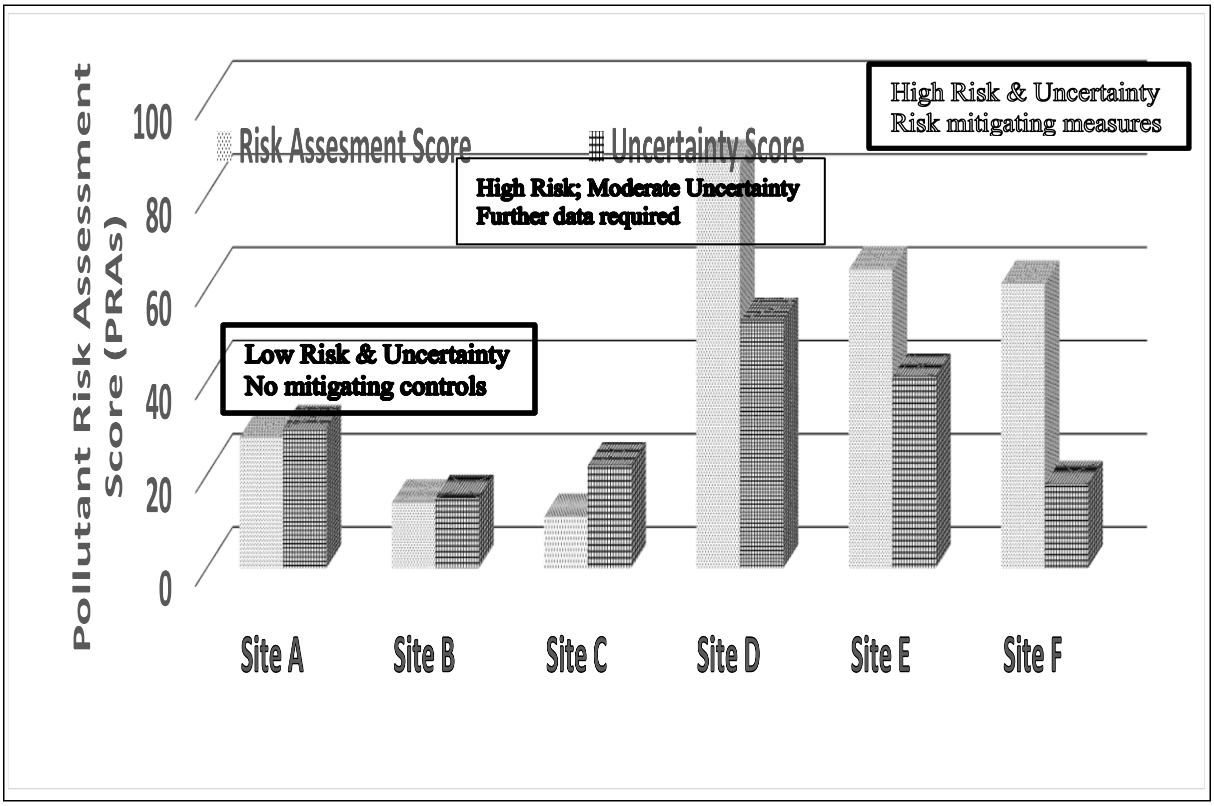

Figure 3.3 Variability in receptor point risk and uncertainty

Table 3.3 and Figure 3.2 suggest that aquifer vulnerability will be a function of multivariate factors of which perhaps the most significant include:

- the physiochemical characteristics of the overburden,
- land use activity and intensity (e.g traffic volumes/types),
- soil and unsaturated zone types/conditions/geochemistry,
- the soil and aquifer media,
- the spatial/temporal characteristics of recharge (including storm event and preceding conditions),
- groundwater yield/abstraction rates

The tiered approach provides a logical process for risk assessment and management as it facilitates the application of optimised data and information relating to the above factors and is focussed on a priority-based assessment of risks. There is often a lack of data at many sites and therefore it is difficult to maintain consistency between risk assessments performed on different parts of the same model or at different sites.

Phased risk based approaches are very common in Europe for urban land and groundwater management but there are many differing modelling approaches employed by national agencies and groups with results varying by orders of magnitude, particularly where parameter default values are used to benchmark the methodological application. The organisation of the methodology into a multilevel hierarchical structure does help to reduce the effect of data loss; allocation of individual risk scores can be conducted separately (Working Group on Groundwater, 2005). Spreadsheets, flow chart analysis and vulnerability mapping (including source protection zonation), support the first initial screening tier with progressively more complex theoretical/conceptual modelling codes supporting the higher risk assessment tiers. Some of the working assumptions and critique of pollutant transport and fate modelling methods can be found in McMahon et al (2001c), Worall and Kolpin (2003), Reilly and Harbaugh (2004) and Troldborg (2010). Hariharan and Shankar (2017) provide a detailed critique of MODFLOW which has well recognised deficiencies related to grid definition and connection as well as issues related to limited representation of anisotropy with hydraulic conductivity always being perpendicular to the grid cell faces. This causes major limitations when dealing with fractured rocks such as the confined UK chalk aquifer.

Perhaps the primary modelling challenge arises from the complex mix of hydrologic/hydraulic factors with lithological and geochemical fate/transport parameters as well as accurately assessing the reality of the modelling setup. It must be remembered that the majority of contaminants do not move at the same velocity as the groundwater and the rate of migration is locally limited by interaction with the aquifer e.g through cation exchange or sorption to organic matter etc. The processes of dispersion, attenuation and degradation of pollutants are dependent on the type/nature of the individual pollutants and the aquifer properties and this will vary both in time and space. However, contaminants such as bromide, which travels at comparable velocity to the groundwater flow, can make good tracer compounds and can be used to calibrate the groundwater model.

However, it is clear that a sound understanding of the supporting models and their working assumptions is vital for the appropriate selection and application of the software tools (Whittaker et al, 2001). Existing aquifer classifications are principally focussed on the significance of the aquifer as a water supply resource and there is limited emphasis on the role of groundwater in terms of supporting surface water baseflows or aquatic ecosystems. Such considerations may require a re-classification and revised tools to undertake the analysis of such matters as groundwater-surface water interactions, stratification characteristics and recharge estimations for infiltration drainage, all of which could be helpful to the management of highway drainage and receiving waterbody protection. There is undoubtedly considerable uncertainty (and variability in practice), regarding the impact of SUDS infiltration systems on aquifer recharge and in the delineation and design of the drainage field. This is important as elevated hydraulic loadings, such as can occur during extreme storm event drainage, could lead to bypassing of the soil zone and result in significant reduction in attenuation capacity within the underlying unsaturated zone. Lysimeter studies of highway runoff disposal via infiltration –type SUDS controls suggests that they provide highly effective pollutant entrapment and biodegradation treatment (Jefferies et al., 2008). In these respects the technology of surface water drainage (including highway runoff), is somewhat ahead of the planning and regulatory regime. However, these studies have been time limited (<200 days), not conducted on soil and bedrock materials possessing preferential pathways or tested under extreme event conditions.

There is an understandable emphasis on the protection of groundwater supply sources in all national management strategies but urban/highway groundwater behaviour is still relatively poorly understood and the subsurface biodegradation pathology and fate of urban/transport sourced contaminants has only a limited research and data information base. There is also a need for the potential threats that degraded urban/transport derived groundwater might have on aquifer (and surface water) ecosystems and the timeframe over which any impacts might occur. It may be that monitored natural attenuation (MNA) could be a cost-effective means of managing urban/transport derived groundwater contamination, at the very least for low trafficked highway locations. MNA could be regarded as representing a pathway blockage "technology" but little guidance exists on its practical implementation as a potential management tool.

3.2.4 Characterisation of parameter factors influencing groundwater pollution potential

The risk-based characterisation of both surface and groundwater bodies required under Article 5 of the EU Water Framework Directive (Directive 2000/60/EC; 2000) as part of River Basin Management Plans includes reference to the pressures and sources resulting from urban and transport activities. The potential risk vulnerability of individual groundwater bodies was essentially assessed from the thickness and permeability of the surficial overburden (or drift deposits) together with some reference to aquifer yield. The unsaturated (vadose) zone is not relied upon for attenuating pollutants since many bedrock formations possess secondary rather than primary porosity. Four categories of vulnerability or risk were recognised in the UK, Irish, Dutch and other EU member states methodologies based on their permeability characteristics. Table 3.4 illustrates the vulnerability mapping guidelines produced for Ireland (Daly and Misstear, 2001) which were adapted from UK guidance although it must be noted

Table 3.4 Vulnerability mapping guidelines

Vulnerability Rating	**Hydro-geologic conditions**				
	Subsoil Permeability and Thickness			**Unsaturated zone**	**Karst**
	High permeability (e.g. sand/gravel)	Moderate permeability (e.g. Glacial till, loam)	Low permeability (e.g. clay silt, clay, peat)	Sand and gravel aquifers only	(<30m radius)
Extreme (E)	0 – 3.0m	0 – 3.0m	0 – 3.0m	0 – 3.0m	-
High (H)	>3.0m	3.0 – 10.0m	3.0 – 5.0m	>3.0m	N/A
Moderate (M)	N/A	>10.0m	5.0 – 10.0m	N/A	N/A
Low (L)	N/A	N/A	>10.0m	N/A	N/A

that hydrogeological conditions in other EU locations may well differ from those listed in the table. Massive fractured bedrock such as typifies the karstic areas of Portugal or Slovenia will fundamentally modify the characteristics of the unsaturated zone such that vulnerability ratings become high (or extreme) throughout the full depth and radii from the pollution source. The four rating subdivisions can be shown on a groundwater vulnerability map or can be assessed on a site by site basis. Figure 3.4 shows the distribution of the designated extreme vulnerability groundwater distribution within Ireland based on this characterisation methodology. The guidelines were accompanied by default values for groundwater body thickness pertaining to differing generic aquifer types and yield characteristics as a basis to define boundary conditions (UKTAG, 2012). Figure 3.5 is an extract from the EU Article 5 submission made by the UK for the Thames Basin illustrating the high risk potential posed by the fractured lithology of the Chalk aquifer and which demands that transport authorities undertake high tier detailed risk assessment for any drainage infrastructure intended to be directed to sensitive groundwaters (Highways Agency, 2009a).

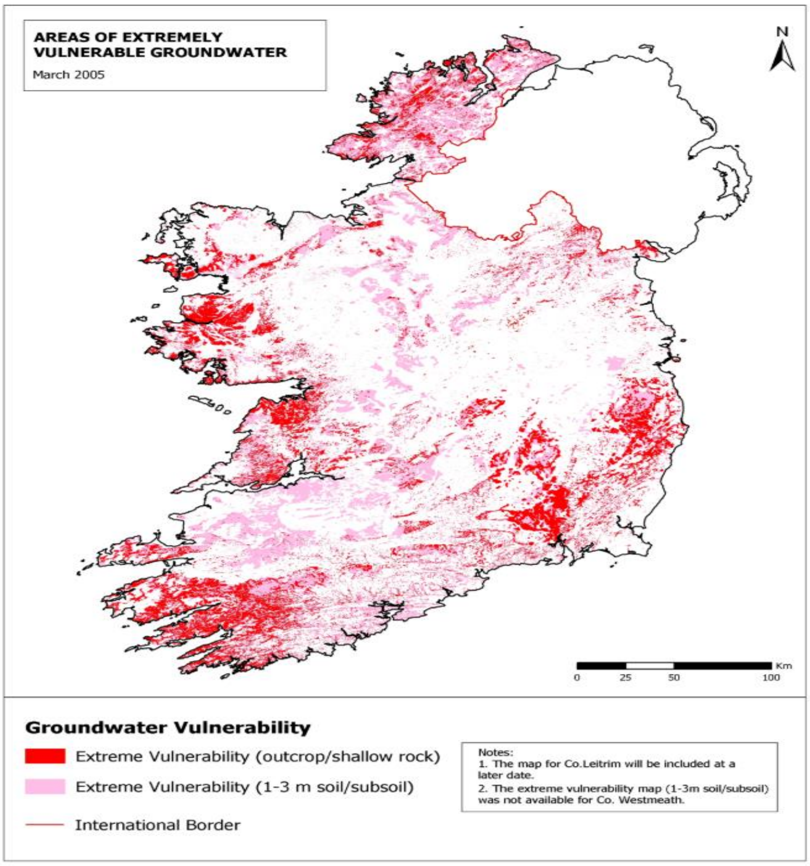

Figure 3.4 Extreme groundwater vulnerability for Irish aquifers

The maps provide a spatially explicit assessment of the current groundwater status at an areal/catchment level and help identify the level of risk vulnerability and priority areas which require further data/information. Such vulnerability distribution maps provide a valuable initial screening tool for preliminary planning purposes and are available at a scale of 1:100,000 to accompany geohydrologic groundwater contour maps to facilitate the determination of parameters such as hydraulic gradient, likely flow paths, lithological controls etc.. The European Environment Agency (EEA) holds the WISE WFD Database (www.eea.europa.eu) on which member states have lodged their national inputs including details of groundwater status, potential and pressures/impacts such as derived from transport activities.

The most common variables generally used to characterise the risks associated with groundwater vulnerability to pollutant discharges from surface sources such as highway drainage include: net recharge, soil properties (type, infiltration rate, field capacity, permeability etc..), unsaturated zone lithology and thickness; depth to groundwater; aquifer media properties (e.g. bulk density, effective porosity, saturated thickness etc..) and aquifer hydraulic conductivity. Other frequently used factors include ground slope (or topography as termed in the DRASTIC model) overburden and plume thickness, dispersivity and solubility/toxicity as well as degradation rates etc.

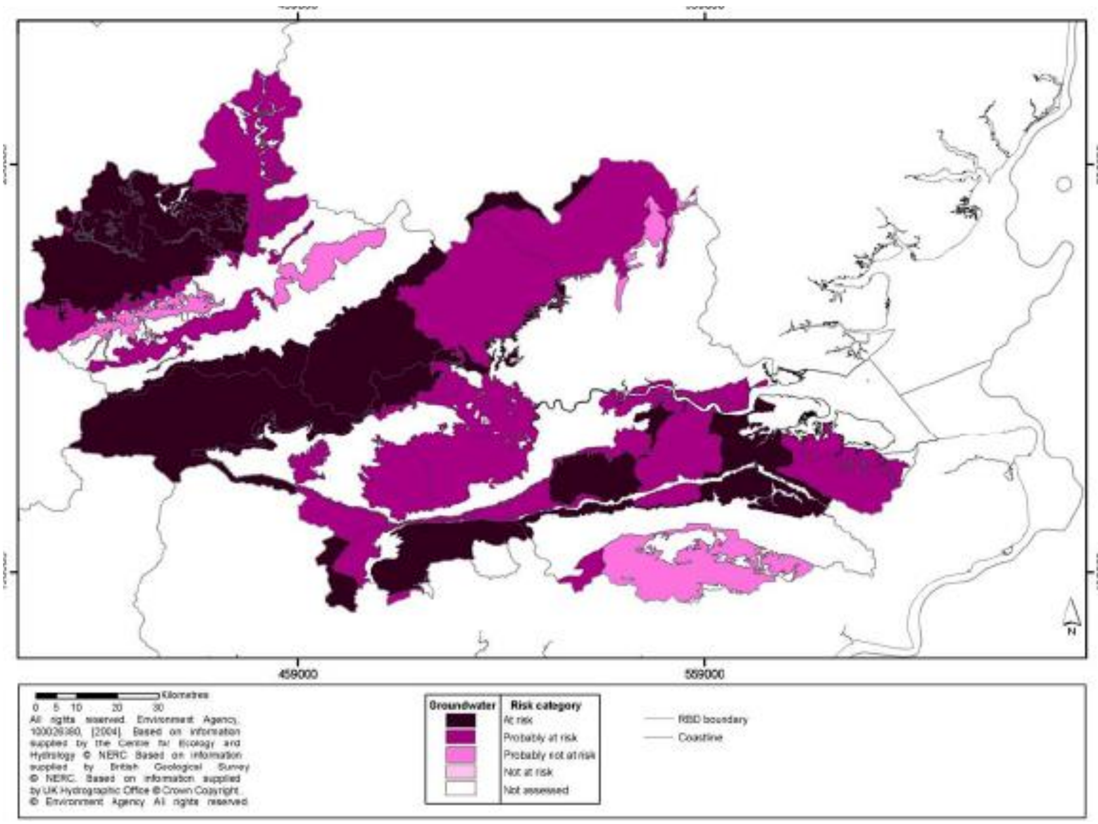

Figure 3.5 Groundwater at risk vulnerability in the Thames Basin UK

Table 3.5 illustrates the matrix characterisation adopted within the DRASTIC intrinsic vulnerability (IV) methodology where the IV index is based on the benchmark rating of seven input parameters (indicated by the bold red letters in the first column which spell out the DRASTIC acronym) and which are considered to control the vertical migration of potential pollutants down into the aquifer. The table also shows (in blue bold type), the parameters which are used to characterise the principal six factors adopted by the UK highways drainage manual (Design Manual for Roads and Bridges, DMRB) for the risk assessment of disposal of highway runoff to ground (Highways Agency, 2009b).

There is an additional seventh factor in the UK DMRB method not shown on the table which refers to the Source Control type and design geometry intended to reflect the effectiveness and efficiency of the specific mitigating device/system installed to reduce potential highway runoff impacts. Thus a grass swale channel is considered the least effective and should be restricted to traffic densities less than 5,000 AADT (given a rating of **1**), soakaways/lagoons for 5,000 – 50,000 AADT (rating **2**) and large infiltration basins/field soakaways for >50,000 AADT (rating **3**). An overall weighting of **15** was allocated to this parameter. This seventh factor in the DMRB approach is not an intrinsic influencing process or property variable being an anthropogenic-sourced mitigating drainage infrastructure interposed between the contaminant source and subsurface flow pathway. The relative efficiency and effectiveness of the control device will be dependent on the design, construction and maintenance standards implemented for the drainage device. Thus the technology-based benchmark ratings and weightings are very different in character from the other intrinsic parameters contained in the risk assessment procedure. It is included in the DMRB matrix partly because the HAWRAT surface water risk assessment model (Highways Agency, 2009b) allows the user to exercise a first-order treatment to the drainage runoff concentration prediction prior to onward transport to the receiving waterbody.

Table 3.5 DRASTIC and DMRB model parameter characterisation

Parameter (and unit)	Impact on IV index	Benchmark ratings		Parameter weighting	Comment
D Depth to water (m) (3)	Increases migration pathway for surface pollution to reach WT and aquifer	0 – 3 3 – 6 6 – 9 9 – 12 12 – 15 15 – 18 18 – 21 21 – 24 **(<5)** 24 – 27 **(15 – 5)** 27 – 30+ **(>15)**	**10** **9** **8** **7** **6** **5** **4** **3** **2** **1**	**5** **(20)**	Shallower the water depth, the greater is the potential risk to the aquifer. Index of depth to the vulnerable unsaturated zone.
R Recharge (mm/y)	High head promotes vertical migration through the vadose zone	0 – 51 51 – 71 71 – 92 92 – 117 117 – 148 148 – 178 178 – 216 216 – 235 235 – 254 254+	**1** **2** **3** **4** **5** **6** **7** **8** **9** **10**	**4**	Defined as: Net Recharge = Precipitation – Evapotranspiration – Runoff. Factor (2) replaces the **R** parameter in the UK DMRB method
A Aquifer media (4)	Permeable (and fractured) rock materials render aquifer more vulnerable	Karst Chalk Basalt/Fluvioglacial Sand & Gravel Massive Sandstone Massive Limestone Bedded sst/lst Glacial till/Organic Metamorphic/Igneous Massive shale/Clay	**10** **9** **8** **7** **6** **5** **4** **3** **2** **1**	]](3)]] **3**] **(20)** }(2) } }(1)	Flow type as determined from rock properties in the UK DMRB method Pollutant migration characteristics need to be defined
S Soil media	Thin and permeable soils promote high infiltration rates to groundwater	Thin/Absent Sand Peat Clay	**10** **9** **8** **7**	**2**	Ranging from Extremely poorly drained (**1**) to Well drained (**7/8**) up to Very rapidly drained (**10**).

		Loam Sandy Clay Silt Clay/Silt Loam Silty clay Non-shrinking clay	**6** **5** **4** **3** **2** **1**		Might be more appropriate to use Infiltration rate as the key factor
T Topography (Slope) (%)	Steeper slopes likely to lead to higher runoff volumes and rates.	0 – 2 2 – 6 6 – 12 12 – 18 >18	**10** **9** **5** **3** **1**	**1**	
I Impact of vadose zone **(6)**	Low permeability may impede infiltration and pollutant transfer	Clay/Marl Sand/Silt/Till Bedrock Glaciofluvial Gravels Karst	**1** **4** **6** **8** **8** **10**	(1) >15%clay (2)5 – 1%clay (3)<1%clay **5** **(7.5)**	Need to define influence of biodegradation, immobilisation and/or decay on key pollutants
C Hydraulic conductivity (m/d)	High conductivity leads to more rapid and higher diffusion	0.1 – 3.0 3.0 – 5.0 5.0 – 8.0 8.0 – 10.5 10.5 – 14.0 14.0 – 20.0	1 **2** **4** **6** **8** **10**	**3**	
Traffic Density (AADT) **(1)**	Influences source pollutant concentrations and loadings	>50,000 15 – 50,000 <15,000	**(3)** **(2)** **(1)**	**15)**	
Storm Event Rainfall **(2)**	Volume (mm/y)	>1060 40 – 1060 <40	**(3)** **(2)** **(1)**	**(15)**	
	Intensity (mm/hr)	>47 35 – 47 <35	**(3)** **(2)** **(1)**	**(15)**	
Porosity (%) **(5)**	Controls matrix storage and flow capacities and properties	Fine sand and below Coarse sand V. coarse sand and above	**(1)** **(2)** **(3)**	(7.5)	Effective grain size based on lithology

The sum total of weightings in the UK DMRB methodology is 100 whilst the DRASTIC weightings sum to 23. Whilst the benchmark rating allocations can be arguably justified in terms of the range of values/information available or observed for the respective individual factorial parameters, the parameter weightings are essentially based on more subjective expert judgement. It is not clear for example, how or why the parameter weightings are derived e.g. what is the reason that aquifer porosity is considered to carry half the weighting of the AADT parameter in the DMRB methodology? The rating and weighting values allocated to various parameters varies considerably between differing national methodologies as does the specific range of values allocated to the benchmarking of the ratings which depend on local circumstances such as total recharge/rainfall depths, slope gradients, groundwater depths etc.. There may be some suggestion in the benchmark ratings shown in Table 3.5 that there is a steady progressive risk associated with each of the factor variables. However, there is no "built-in" linearity in the risk rating scores and some may well be exponential in form in terms of their relation with time and/or distance from the pollutant source. It may be that in some cases the data will not be available for some of the characterising parameters or simply may not be readily adopted into a usable risk assessment matrix. The basic working assumption is that the influencing factors should be derivable from either field or other validated sources or from some accessible generic source.

The final vulnerability is calculated by summing the spatial distribution maps for each parameter according to the specific ratings and weightings assigned in the approach. For the Drastic index illustrated in Table 3.5 the DRASTIC IV is:

IV = 5D + 4R + 3A + 2S + 1T +5I + 3C

whilst for the DMRB index:

IV = 15(1) + 15(2) + 20(3) + 20(4) +7.5(5) + 7.5(6) + 15(7)

Other parameters of significance for the characterisation of groundwater pollution potential can include:

- hydraulic gradient (m/m) which is very site specific and based on levelling of groundwater wells. It has variable relationships with groundwater level and receiving waters due to rainfall and seasonal conditions
- effective aquifer porosity (%) which is also site specific and difficult to measure and especially so where a varied lithology occurs.
- bulk density (g cm^3) of soil and aquifer zones
- organic carbon (OC) fraction and soil/water partitioning which effects the pollutant adsorption and exchange capacity and which is quite frequently used in "fingerprinting" benchmark values for hydrocarbon concentrations allowable in higher tier assessment of highway runoff (Table 3.6).

Table 3.6 Characteristics of hydrocarbons in highway runoff (After Ellis et al., 1997)

Hydrocarbon	Target concentration (µg/l)	Henry's law constant	OC partitioning coefficient (K_{OC}; l/kg)	Degradation half-life (days)	K_{OC}/Kd Ratio
Aliphatic C_{12-16}	300	1.71E+03	5.37E+06	201	5.01E+04
C_{16-21}	300	1.07 E+03	6.37E+08	920	6.31E+08
Aromatic C_{12-16}	90	1.26 E-02	5.01E+03	204	5.01E+03
C_{16-21}	90	6.95E-04	1.6E+04	920	1.58E+04
Fluoranthene	6.0063	6.29E-05	1.82E+04	880	1.23E+06

- aquifer geochemistry in and above the aquifer to determine the degradation conditions for various pollutant species in both sorbed and dissolved phases (single or multi-phase flow). The degradation rate of various pollutants can help combine vulnerability data with knowledge of potentially contaminating land areas. The degradation parameter is quite sensitive to the input data and changing the half-life or distance to compliance of a particular pollutant for example, can result in substantial differences in the Remedial Target Values (RTVs).
- dispersivity in longitudinal, transverse and vertical directions within the aquifer
- receptor type/location/status: abstraction, springs, surface water, wetlands etc..
- pollutant plume thickness which can be assumed to be equal to the saturated aquifer thickness.
- well capture zone as defined by particle tracking analysis and distance to the compliance point from the pollutant entry point.

Review of the literature presented here would suggest that majority attention has been focussed on three principal influencing controls on subsurface pollutant migration from highway drainage systems:

- net infiltration rate (per unit area of the unsaturated zone)
- the influence of sorption phenomena
- the influence of biodegradation, decay and immobilisation

Whilst this focus can be accepted, it must be appreciated that these factors are not readily accessible or benchmarked. It is possible to use surrogate factor values, for example the percentage clay or organic carbon content as a proxy for the sorption and retardation pollutant rates, but the level of uncertainty associated with the resulting prediction must render it subject to considerable caution. Additionally, treating all contaminants as non (or very slowly) degrading will result in an overly conservative assessment and yield an overestimation of risk.

Figure 3.6 provides an outline summary of the structure, stages and characterisation procedures that might be contained in detailed risk assessment of groundwater bodies from potentially prejudicial highway discharges based on the application of the Source-Pathway-Receptor (SPR) model. As shown in Table 3.7, data requirements for a high tier risk assessment will be substantial with increasing detail demanded at each successive tier of assessment.

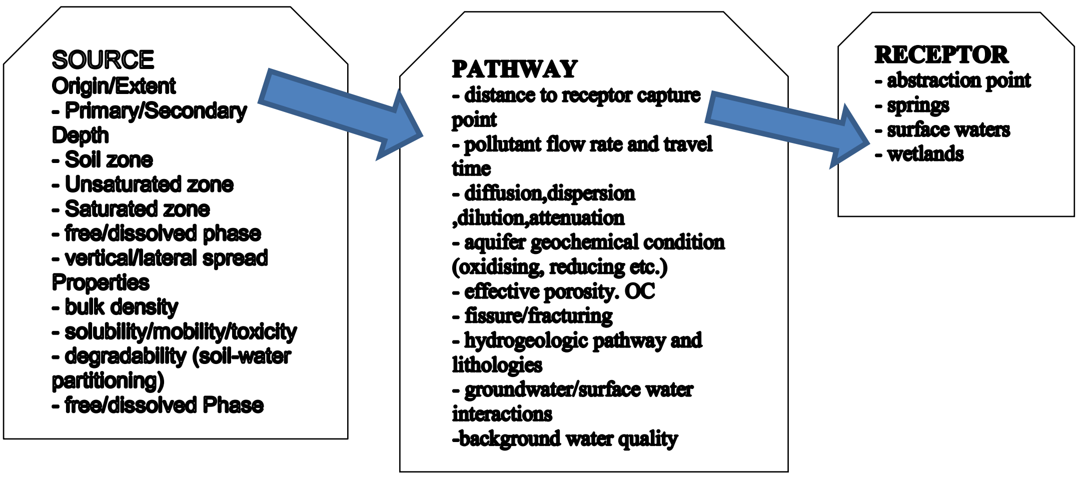

Figure 3.6 SPR risk assessment

At the initial preliminary Tier 1 phase, a general qualitative assessment is only required with influencing factors being primarily (if not exclusively) derived from existing data sources such as soil/surficial, geologic and hydrogeology maps, geological memoirs and utilities reports, borehole records and local trial pits with only a bare minimum of site investigation data inputs. Tier 1 analysis might only be focussed at assessment of soil/surficial layer characteristics with investigation of groundwater factors commencing at Tier 2 (Table 3.7).

Table 3.7 SPR data analysis requirements

Tier 1	Tier 2	Tier 3	Tier 4
Soil Assessment - porewater concentrations - total soil concentrations - porosity - bulk density - partition coefficient (OC, K_{OC}) -pollutant properties e.g. solubility, density etc.	- hydraulic conductivity - hydraulic gradient - saturated depth -pollutant concentration in plume - groundwater abstraction rate	As for Tier 2 plus: - distance to compliance point - pollutant degradation characteristics - partition coefficient in saturated zone - depth of saturated zone - porosity/dispersivity - saturated bulk density - sorption characteristics	As Tier 3 plus: - hydraulic conductivity - contaminant distribution - degradation rates - aquifer geometry - porosity/storage capacity - groundwater heads (e.g. changes with time) - abstraction/recharge, spring discharges etc.

Examples of the lower Tier 1 groundwater risk assessment for highway drainage sites are given in Annex II, Method C of the DMRB (Highways Agency, 2009b) which uses a simple qualitative approach to derive a vulnerability index score. Data sources are essentially map based with input from validated local or national organisational sources e.g. traffic densities, rainfall statistics etc..

4. Conclusions

This report builds on the findings of D1.1 and D2.1 to develop a long-list of parameters which have the potential to influence the impact of highway discharges on receiving water quality. Using a risk assessment framework, parameters are categorised into broad groupings which list a diverse range of aspects/processes/activities identified in the literature as having the potential to influence the:

- Likelihood of highway runoff discharging at a concentration that may cause harm during:
    - road operation (see Table 2.1)
    - construction (see Table 2.2a)
    - maintenance (see Table 2.2b)
- Magnitude of impact of a hazardous discharge in terms of the:
    - vulnerability of surface waters (see Table 3.1)
    - vulnerability of groundwaters (see Table 3.3)

As well as listing the parameters themselves, the nature and type of influence of each parameter is described and an indication is provided of how changes in the parameter can increase - or decrease - highway pollution loadings (likelihood of occurrence) and vulnerability of receiving waters (i.e. magnitude of impact).

The next stage of the research in Task 2 of the PROPER project will involve assessing the relative importance of the identified parameters with a view to developing a short list for use within the Decision Support Tool (DST) planned for D2.5 (i.e. the criteria and supporting indicators). Following the approach set out in Raggett et al., (2009), this short-listing process will involve a screening of all parameters in relation to their:

- feasibility i.e. is sufficient data available to enable the parameter to be used systematically on a local, regional or European scale? If not, although inclusion of a parameter may be desirable its inclusion may not be feasible. However, identification of this lack of data is useful in establishing data needs to inform future research agendas
- scientific soundness – data quality is an important consideration in ensuring that interpretations are scientifically sound e.g. data sets should be as recent as possible if parameters are to be used in an international context (as is the case in this work) and they should be designed such that there is an appropriate level of comparability between countries.
- mutual independence – the ability to assess one parameter should not depend on the assessment of another parameter
- operationality – is the parameter applicable/relevant to all types of water bodies that may be considered?
- non-redundancy – there are two types of redundancy which should be screened out during short-listing:
    - all short-listed parameters should be necessary, important, and not duplicated
    - any parameters against which all water bodies would score identically should be discounted from the analysis as the scoring will not affect the results

In practice, total feasibility, operationality, non-redundancy etc. may not be entirely achievable. For example, the quality of available data may be poor or incomplete, high quality data may not be of operational use and the available data may not be available in formats that are comparable across national borders. An important challenge is therefore to avoid an endless search for perfect data and to instead use existing databases as fully as possible and to be creative in the approach to data handling. The results of this initial parameter screening approach will be discussed with IAB members and their comments/preferences taken into account in the final criteria/indicator selection process.

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