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Salt SMART
Environmental damages caused by road salt -a literature review

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Summary

The literature review focus on environmental damages on surface water (flora and fauna), ground water, terrestrial plants and soil caused by using deicing chemicals in winter road maintenance . An evaluation of the most actual alternatives to sodium chloride is given. An overview of how to handle salt storm water from roads are also given.

Key words

Road salt, deicing chemicals, soil, surface water, groundwater, biological effects

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Preface

There have been numerous investigations into the use of road salts and their environmental consequences. This literature review contains both national and international literature focusing on the effects road salting has on surface water (flora and fauna), groundwater, terrestrial plants and soil.

Bioforsk completed this review in cooperation with UMB (University of Life Sciences, with Bioforsk being mainly responsible). Researchers from Denmark, Sweden and Finland formed a Nordic reference group. They produced a temporary draft of the review by providing both knowledge and literature from Nordic countries.

Results from the literature review are found in two reports:

The main report (Teknologirapport no. 2535) refers to and summarises and concludes the literature used. It also presents suggestions for further research within the four main themes of the report: surface water, groundwater, flora and fauna and the handling of surface water plus other measures against the dispersal and the effects of road salts.

Reference report (Teknologirapport no. 2540) presents all the abstracts or summaries of the literature. These are presented alphabetically and therefore found in the same order as the reference list in the main report.

Carl Einar Amundsen at Bioforsk has been the main responsible person for the setting up of the literature review position. Jørn Arntsen and Kjersti Wike have been the contact people at the Norwegian Public Road Administration (Statens vegvesen). The report has been produced in full by Bioforsk.

Contents

1. Summary

Background
This literature review focuses on the environmental damage road salt (Sodium chloride) can cause in surface waters (flora and fauna), groundwater, vegetation (natural and cultivated plants), and soil. The most actual alternatives to sodium chloride used to de-ice the roads are considered. In light of the literature that exists, an assessment is given for estimations of critical load for species and natural environments. There is also a description of how road salt run-off water can be managed.

Assessment of de-icing chemicals
To main groups of de-icing chemicals are considered in this report: chloride based and organic based. Among the chloride based chemicals, sodium chloride, calcium chloride and magnesium chloride are reviewed. The organic based agents considered include sodium acetate, calcium acetate, calcium magnesium acetate, calcium formate, sodium formate, propylenglycol and urea.

Main Results

Soil
Sodium chloride alters the soil structure and composition of cations during ion exchange in soils and soil solutions. This can in turn result in variation in pH in soil and soil solutions, as well as changes in the biogeochemical cycle to pollution of soil (for example an increased mobility and availability in heavy metals). The washing out of calcium and magnesium from the soil as a result of the sodium chloride use during winter maintenance, leads to increased potential of colloidal transport in soil, with a possibility of reduced hydraulic conductivity if pores become blocked with particles. Increased mobility of organic and inorganic colloids result in an increased mobility and wash out of heavy metals such as lead and copper from polluted surface soils, whilst the addition of chloride will increase the mobility of amongst others cadmium and zinc due to the formation of release of chloride complexes.

Calcium magnesium acetate contains calcium and magnesium both of which stabilise clay particles and improve drainage and air access in the soil (generally a better soil structure). The advantage with using calcium magnesium acetate instead of sodium acetate or potassium acetate is that calcium and magnesium bond stronger to soil than sodium and potassium and gives a lower leakage into groundwater. High levels of calcium can temporarily result in a reduced availability of magnesium and potassium to plants.

Degradation of acetate and other organic based de-icing agents can result in lack of oxygen in soil because oxygen is used during degradation. This can lead to increase in transport of iron and manganese because precipitated oxidised binding of iron and manganese is reduced, released and becomes more mobile. This can also result in an increase in release of heavy metals from soil.

To soil it is more relevant to produce a sensitivity index or hazard index rather than critical load. In such situations it is necessary to understand the connection between roadside soil types, characteristics of these (for example clay content) and eventually changes in composition over time. This type of criteria can be determined by the amount of sodium in the ion exchange capacity and the clay content in the soil. On stretches of road where sodium chloride is added periodically and which have high clay content, the risk will be greater for negative effects on soil.

More detailed knowledge on the relationship between the use of de-icing agents on roads and the effects on flora and fauna is necessary to develop critical load for soil.

Analyses of soil chemistry (total and dissolved) along the roadside should be undertaken at a higher level because this allows a difference between short and long time effects of de-icing agent use during winter maintenance (direct spray vs. growth conditions in soil). Changes in soil characteristics over time with the use of alternative de-icing agents are researched little in Norway. These types of investigations would be useful when recommendations to alternative agents during winter maintenance are developed.

Microorganisms and fauna in soil
Microorganisms in soil close to high trafficked roads can be damaged by sodium chloride. In the soil ecosystem micoorganisms, flora and fauna is dependant on each other and it is unclear how this ecosystem is affected by different de-icing agents or a combination of chemicals.

Chronic effects on soil springtails (Collembola) are proven at concentrations as low as 480 mg sodium chloride (or 280 mg Cl/l), whilst earthworms (Lumbricidae) have been shown to tolerate more sodium chloride. As for aquatic organisms both chronic and sub lethal effects occur by salt concentration that are considerably lower than the concentrations where acute effects occur. By carrying out effect studies it is therefore important that chronic and sub lethal effects are recorded and not only acute.

It is unclear where the critical load for chloride is for soil fauna.

If degradation of organic de-icing agents is based on the soil infiltration (and not the accumulation and cleaning in ponds) it is important to investigate which soil types will be suitable as a refining material and which are not suitable for this task. Combinations of de-icing agents which are both organically based and chloride based can result to a poor decomposition capacity in soil of organic chemicals. Such possible interaction effects should be investigated further.

Plants
The registration of salt damaged plants found on Norwegian roadsides and those found in the laboratory show large differences in the tolerance of Scots pine (Pinus silvestris), Norway spruce (Picea abies) and silver birch (Betula pendula) when it comes to salt uptake through the roots. All three species are however particularly sensitive to salt spray. Spray damage to Scots pine and silver birch is common. However root uptake rarely causes huge damage. Among a number of grass species that are usual along roadsides the following ranging is done; perennial ryegrass (most tolerant) > red fescue > creeping bent > hard fescue > Kentucky bluegrass >sheep’s fescue > common bent (least tolerant). Also amongst these grass types there are differences between sorts and species. Also amongst perennials (for example Geranium) it is shown to be a considerable difference between sorts and certain sorts should not be used even in areas with a more moderate sodium chloride exposure.

In numerous studies (including Norwegian) it is shown that the deposition of air borne sodium chloride declines exponentially with increasing distance from the road. The greatest spray damage is usually found within a zone of around 10 m from the road, for some species more. The most of this spread comes from run-off and not by the air. Forests along roads are proven to be effective in catching the salt spray and in this way reducing further spreading from the road. Variation in local soil and climate determines the magnitude of salt damage to vegetation.

Changes in species composition along roads is a natural result of the use of sodium chloride and studies show that Norway spruce can be out competed along certain roadsides. Alternative de-icing agents have also effects on plants, but more studies imply that the effects are less than when using sodium chloride. There is a need for more detailed documentation of these effects before clear recommendation regarding alternative de-icing agents on Norwegian roads.

In city centres the speed levels are low and risk of spray damage is low. Salt damage as a result of uptake through soil is temporarily a serious problem. It can take a long time before symptoms of salt damage cease. This is believed to be because of delayed leaching of salt due to solid deck such as asphalt and stone.

The critical load for salt can be identified by relating growth disturbance or extent of damage symptoms to the concentration of chloride or sodium in the leaves, soil, soil solution and eventually concentration in irrigation water or critical load can be related to dosage of a certain amount of sodium chloride to soil.

It is a problem to determine critical loads that are applicable in a certain field situation because damage extent is dependant on a number of factors. Symptom development and concentration changes in plant weave can take place differently over time and relationship between them will therefore be dependant on time of sampling. Sodium chloride damages plants both through uptake through the roots and directly through spray and there isn't necessarily a link between tolerance through roots and direct spray.

It is strongly suggested that plants that tolerate raised salt levels and that the choice of plant type takes into consideration which exposure dominates the area. There is a need for more documentation of the effects and selection of salt tolerant plant types that are used along salted roadsides and documentation of variations in damage because of salt spray.

Surface water

Lakes in Norway which are affected by the use of sodium chloride during winter maintenance have developed salt gradients (difference between water in the upper level and bottom level > 10mg C/l). In the bottom levels of salt affected lakes, oxygen depletion gives higher concentrations of iron and manganese in the water. Use of chloride salts can lead to an increase in concentration of heavy metals and base cations (such as Ca and Mg) in the surface water.

Addition of sodium chloride in soils leads to a higher ion strength in the soil solution which in turn gives a reduced release of humus and therefore a lower humus content in water. This results in worse buffer systems and lakes light and temperature ratio can alter.

Chloride concentrations in Norwegian lakes are usually between 1-10 mg/l, even if the coastal surface waters can have higher concentrations (30 mg/l). Concentrations of salt from road run-off can reach up to 10 grams per litre.

Acute effects of exposure < 4 days, acute effects of exposure 1 week and chronic effects arise at Cl-concentrations at respectively (ca.) 6000 mg/l, 1100 mg/l and 560 mg/l if it allows effects on 50 % of organisms (EC_{50}) (fish, shellfish, algae). If effects are based on only 5 % of organisms then chronic effects can arise at concentrations of ca 200 mg/l (based on Canadian studies). Changes in species composition and physiological changes at certain species (chronic effects) arise in other words with much lower concentrations than acute effects. The general trend in a number of studies shows that the chloride based de-icing agents are less toxic to aquatic organisms (fish, shellfish, and algae) than those based on acetate. Sodium chloride is shown to be the least poisonous agent among the chloride based, whilst CMA is least poisonous amongst the acetate based.

If it is taken into consideration the organic combination (formate, acetate) are broken down in the unsaturated zone in soil, there is less probability that these will cause negative effects in water. It is, naturally enough, dependant on that the organic de-icing agents are infiltrating the soil and not flowing directly in to the surface water.

International compilation show that the relation between chloride levels in freshwater and acute and chronic effects on fish, shellfish, and algae are relatively well known and critical loads for fish and invertebrates can be established today by going extensively through the data material that exists. However, it must be discussed which organisms and ecosystems are the most sensitive and how much protected is wished for (and economically viable) that these will be. There is considerably much less data available when it concerns the effects of other de-icing agents that sodium chloride and critical loads for these will be determined with less certainty.

Groundwater

Decomposition of organic de-icing agents increases with temperature and nutrient contribution (N and P) increase decomposition rates. It is therefore important to ensure enough residence time in the unsaturated zone to ensure that chemicals are broken down before the melt water reaches groundwater. Summer has higher soil temperature and will probably increase decomposition rates.

Organic de-icing agents have different oxygen consumption, but generally it is like that decomposition of these often result in increased iron and manganese concentrations in groundwater from soil and this can give disadvantages in relation to use for drinking water. Use of organic de-icing agents are also shown to give increased concentrations of among other things magnesium, zinc, barium, calcium and sodium, as well as higher pH levels in groundwater. Decomposition of propylenglycol under anaerobic conditions can give formation of poisonous mercaptan that give off a rotten smell.

A number of models can be used to estimate critical limits under certain circumstances. In addition to content of sodium and chloride, it is relevant to look at the other elements than those used as de-icing agents. The reason is that chemicals will break down to other substance that for example can be measured as TOC, or chemical ionic exchange reactions that wash out other elements from the soil. Iron and manganese are elements that typical appear at higher concentrations where there is a decomposition of organic substances.

To avoid unlucky influence of groundwater, dilution of road run-off is an important effort. To estimate dilution potential for the national road network, can different mass balance models be used together with information on salt addition and excess precipitation (infiltration). Good meteorological data and data about original state are necessary in such estimations. Other efforts can be to avoid infiltration along vulnerable stretches (collection of surface water), choose degradable de-icing agents or combinations of de-icing agents, or manipulate groundwater flow such as vulnerable recipients or installations (wells) don’t receive run-off from the roads.

Run-off water management

Pollution and environmental problems connected to sodium chloride and other de-icing agents can partly be remedied through a recipient adapted surface water management. Accomplishment of management measures requires a completed mapping out of recipients along roads with consideration to vulnerability towards addition of road salt. Since road salt, especially chloride is difficult to remove the most important effort will be (1) dilution and (2) directing away run-off with high concentrations of road salt.

Storm water systems that are built along motorways in Norway have in principle a form that will contribute to the collection of run-off water along longer stretches of road and lead this controlled to spillage (and eventually treatment) at the closest low point. Leading away salty run-off water from vulnerable recipients is used to ensure groundwater resources used for drinking water. In Sweden, Finland and Norway there are examples of different solutions used for the collection and leading away of run-off. Actual solutions for the collection can be (1) kerb stones and gutters along asphalt kerbs and (2) use of tighter membranes in the roadside ditches. The effect of measures increases with the degree of collection.

Arrangement of localized infiltration of salty runoff on the roadside will contribute to a reduction in salt concentrations added to vulnerable becks, ponds and lakes. This will require uncompacted material with certain water conductivity and infiltration can not be used where there is vulnerable groundwater. Solutions for the optimal infiltration in vegetated road ditches are a usual management strategy internationally and also tried out in Norway.

In Norway sedimentation ponds are the most used treatment of run-off before it is released into recipients. Road salt is not removed, but studies show that high concentrations become equalized. Dependant on the formation and dimensions, a salt layer could be formed and affect the retention time and treatment processes in the pond. Relations connected to these processes and which formation and dimensions of sedimentation ponds are the most environmentally friendly favourable with consideration to salt and other pollution is complex, and needs to be researched further.

Dependant of winter climate and snowfall, will snow ploughing and snow melting mean a lot for leakage and transport of salt and other pollution from roads. Selective snow melting will result in salt and water releasable components mainly will follow the first melt water out, whilst particles, PAH and oil components will be freed first at the last snow melt. Vulnerable areas should be protected against salt and pollution through the driving away of ploughed snow and deposited in areas with management measures and few vulnerable recipients. Clearing the road of snow with motor snowplow out to larger areas along roads can be a simple solution to increase infiltration and dilution of road salt in snow.

The use of other de-icing agents is considered in relation to possible efforts connected to run-off water management. Connected to sedimentation ponds it is expected that calcium and magnesium chloride give small changes in conditions compared with sodium chloride, but can increase the flocculation and sedimentation. All the organic de-icing agents have the potential to create oxygen free conditions at the bottom water in stormwater ponds, and especially if salt layers continue over spring and summer. This will contribute to mobilisation of pollution from sedimented mud.

2. Glossary

Water

Groundwater: water held underground in the soil or in pores and crevices in rocks.
Surface water: lakes, rivers, becks and other collections of water that lie open in the day and are in direct contact with the atmosphere.
Freshwater: surface water with a small content of dissolved salts (among them NaCl), in contrast to saltwater and brackish water.
Holomictic lakes: at some time of the year the water in holomictic lakes have a uniform temperature and density from top to bottom, allowing the lake waters to completely mix.
Meromictic lakes: unlike holomictic lakes the layers of waters in meromictic lakes do not intermix (Figure 1).
Monimolimnion: the dense bottom stratum of a meromictic lake, it is stagnated and does not mix with the water above.
Mixolimnion: the upper layer of meromictic lakes, characterised by low density and free circulation, this layer is mixed by the wind. This layer can have a higher salt concentration but is more or less of the same quality as a holomictic lake.
Chemocline: a pronounced vertical density gradient (decreases in salt concentration with increased depth) in a lake.

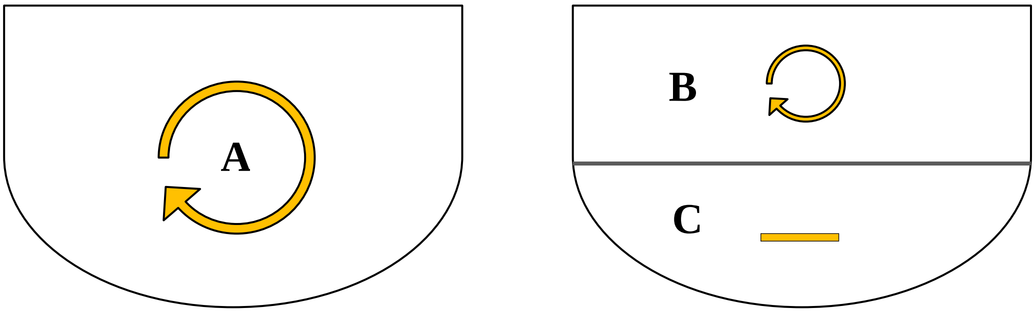

Figure 1: cross section of two main types of lakes circulation patterns. (A) Holomotic lakes where the whole mass of water circulates once or more times a year, plus a merometic lake with a top layer (B – mixolimnion) that circulates, and a bottom stratum (C – monimolimnion) that doesn't circulate (is stagnated). This bottom layer is often enriched with salts and nutrients. C has often little oxygen and can contain relatively high concentrations of carbon dioxide and/or iron, manganese and hydrogen sulphide (toxic). The transitions between B and C have often an obvious chemical divide (a pronounced chemocline).

Critical load (dose terms): often set with a background dose-response test, where effects of organisms are evaluated after the organisms have been exposed to a certain concentration (for example NaCl in freshwater) over a given time period. It is important to separate the dose terms from the concentration terms. In literature the term "lethal concentrations" is often used, but together with a time aspect (for example the trial took place for four days). LC50 (lethal concentration, where 50% of organisms die) must therefore be interpreted such as the concentration that results in half of the organisms die instantaneously or in the long run; LC50 is always connected to a specific time range that will vary between tests. With have attempted to strive for a consistency in this by also stated a timeline. There is another terms in addition, LD50 (deadly dose for 50% of organisms), but here you operate with concentrations per body weight of the organisms – something that is often unpractical in nature (algae, flowers etc). The terms LC50t (deadly concentration for 50% of organisms after exposure over a specific timeline (t)), is also used and corresponds LC50 when a timeline for a test is given. There are standard organisms (ISO standards) that are tested against the exposure of commercial products, but there will always be a deviation between water bodies and standard organisms will often fall outside local interests.

Critical load (lethal vs. harmful): it is important to differentiate a grade of effect; "lethal" is not the same as "harmful". In this context the terms LC50 (see the paragraph above) and EC50 (effective concentration) are much used. EC50 is the concentration of a substance (for example a de-icing agent) that gives a specific effect under test situations after a decided timeline in which 50% of organisms are tested. Here it may be, for example talk about the concentration (over a specified time period) it takes for half of a particular fish species to show visible effects of one kind or another, for example, reduced swimming capacity. EC50 is usually substantially lower that LC50 (LC must be considered as a measure of an extreme consequence, where a proportion of the organisms actually die. One should also be aware that it does not need to be 50% as used in these concepts of critical load, but 50% is often used.

PNEC: predicted no effect concentration. Concentrations lower than those of the PNEC are not expected to have any negative effects.

EC50: concentration (dose) that effects designated criterion (e.g. behavioural trait, growth etc) of 50% of a population.

LC50,20: concentration (dose) that kills 50/20% of the population observed.

Chemical oxygen demand (COD): the amount of oxygen required to achieve a complete chemical oxidation of 1 litre of a sewage sample. Break down often happens with use of KMN04, and now and then K2Cr207 which is a stronger oxidation resource.

Biological oxygen demand (BOD): the amount of dissolved oxygen used by microorganisms to oxidise 1 litre of a sewage sample. BOD is usually measured for 5 days (BOD5) or 7 days (BOD7)

Soil

Cation exchange capacity (CEC, mmolc kg-1): a measurement of the total amount of cations that a soil can bind/adsorb. CEC is usually measured by extracting soil with a solution that contains a large surplus of NH4+ (for example 1M NH4NO3) or Ba^{2+} (1M BaCl2). In natural Norwegian soil it is usually H^{+}, Ca^{2+}, and Mg++ that are the quantitative important cations which are bonded to soil, whilst the amount of K+ and Na+ are lower.

Base saturation (BS %): explains the percentage of *exchangeable* base cations (Ca, Mg, K, Na) in relation to the total amount of *exchangeable* cations that are bound to the soil (Ca, Mg, K, Na, H, Al, Fe).

In connection to high salt levels in soil (both natural and caused by addition of road salt) two terms are used to describe the content of salt in soil and soil solution:

Exchangeable sodium percentage (ESP – defined as (Na/CEC)*100) is an expression to explain how large a percentage of ion exchange complex in the soil is possessed by sodium.

Sodium adsorption ratio (SAR – defined as (Na/(Ca+Mg)1/2, mmolc l-1) is an expression to explain the connection between sodium and calcium and mangnesium in the soil solution.

SHI – salt hazard index = ESP (exchangeable sodium percentage) x clay content in the soil (%). This is an index that can be used to express the danger of erosion as a result of increased content of sodium chloride in soil.

3. Introduction

3.1 Background

There are three particularly adverse situations that may arise in relation to the use of chemicals in winter maintainance on roads: 1) high concentrations of chemicals in the runoff water to lakes, since this may affect lake circulation 2) increased content of chemicals in groundwater, especially drinking water supplies or potential drinking water supplies, 3) ecological effects on aquatic life in rivers and lakes and vegetation and animals in proximity to roads, particularly for vulnerable species.

3.2 Sections and the implementation of the literature study

The literature study focuses on the environmental damage sodium chloride used as de-icing agent can provide the surface water (flora and fauna), groundwater, the vegetation (natural and cultivated plants), as well as in the soil. It provided reviews of the most appropriate alternatives to sodium chloride used to de-icing roads. Based on the literature assessments of tolerance limits for species are given and it is given a description of run-off water management and measures in relation to transport and discharge of road salt.

The report is based primarily on published scientific articles available in literature databases. Literature made available by the Norwegian Public Roads Administration (Statens Vegvesen) was the starting point for comparison, but complementary literature is added from the databases ISI Web, Springer Link, Science Direct, and the like have been used. In addition, the Nordic reference group (chapter 4.3) helped with the literature from Denmark, Sweden and Finland.

Bioforsk (Norwegian Institute for Agricultural and Environmental Research) have been primarily responsible for the literature study, in collaboration with the University of Life Sciences (UMB). The following people have had the main responsibility for the 4 different parts of the study:

1. Surface water: Ståle Haaland (Bioforsk) and Gunnhild Riise (Department of Plant and Environmental Sciences, UMB)
2. Groundwater: Helen French, Bioforsk
3. Fauna and flora: Carl Einar Amundsen, Bioforsk, and Per Anker Pedersen (Department of Plant and Environmental Sciences, UMB)
4. Run-off water management: Roger Roseth, Bioforsk

A Nordic reference group has contributed with suggestions and quality assurance of the literature compilation. It has consisted of:

- Eva-Lotta Thunqvist: KTH (Royal Institute of Technology)/Vägverket consulting
- Morten Ingerslev and Lars Bo Pedersen: Centre for Forest, Landscape and Planning, Copenhagen university
- Jani Salminen: Finnish Environmental Institute (SYKE)

3.3 Readers Guide

It is made to find literature covering the various topics that are of priority in assembling so that important issues have not been excluded. It has been emphasized to use literature that has been subject to quality assurance (peer-schemes) and that are simultaneously published in countries that have comparable climate and natural conditions such as Norway.

The shape of each chapter is different. This is because different people have been involved in the design, the themes are different that the supply of and type of literature within the topics vary. The theme of the surface water management is different from other themes in the sense that there are far fewer studies published in scientific journals, while much of the literature found that reports and brochures where examples of measures are described.

Common to most of the chapters is they are rounded off with discussions and conclusions. In this part of comparison emphasis is placed on the discussion of tolerance limits for sodium chloride and alternative de-icing agents. In the chapters where this is relevant is discussed specifically how the information from literature compilation can be used in relation to the preparation and establishment of tolerance limits.

4. Overview of the de-icing agents

4.1 Introduction

Sodium chloride has been and is the most widely used chemical that is used in Norway and several other countries to prevent and remove ice from the road. Environmental impact, corrosion on road constructions and vehicles has led to increased focus on alternatives to sodium chloride. Roughly these are divided into chloride based products and organic based products (Table 1). Some of these are tested on the road either separately or in combination with sodium chloride. Products such as propyleneglycol is widely used in airports in Norway and other countries, but used very rarely on roads.

The two groups of de-icing agents (chloride and organic based) has two different mechanisms of action. Chloride involves primarily chemical and physical processes, while the organic means as well as involving microbial processes in soil and water. Although the mechanisms of action are different can however the effects in some contexts be the same. This is clearly visible among in chapter 5.

Table 1: Overview of the de-icing chemicals appropriate for use on the road.

Group	Primary constituent	Chemical designation	[1)]Chemical oxygen consumption (COD)/Biological oxygen consumption (BOF) [2)]
Chloride based	Sodium chloride	$NaCl$	0
	Magnesium chloride	$MgCl_2$	0
	Potassium chloride	KCl	0
Organic based	Potassium formate (K-Formate)	$KCOOH$	0.35 / 0.27
	Sodium Formate (Na-Formate)	$NaCOOH$	
	Calcium-magnesium-acetate (CaMg-acetate)	$CaMg (CH_3COO_4)$	1.07 / 0.7
	Sodium magnesium acetate (NaMg acetate)	$NaMg (CH_3COO_3)$	
	Potassium acetate (K-acetate)	KCH_3COO	
	Sodium acetate (Na-acetate)	$NaCH_3COO$	
	Mono-propylenglycol (MPG)	CH_3CHOCH_2OH	1.69 / 0.9
	Fructose / glucose / sodium chloride	$C_6H_{12}O_6$ / $C_6H_{12}O_6$ / $NaCl$	
Other	Urea	$(H_2N)\ _2CO$	213[3)] / 2.0

[1)] Oxygen demand by complete decomposition of 1 mg / l deicing agents. [2)] g COD, BOD / g de-icing chemical, [3)] 2.13 is the theoretical oxygen consumption by decomposition of NH4.

A brief description is given below of de-icing chemicals that are listed in Table 1 and are considered to be the most important for road winter maintenance in Norway.

4.2 Chloride based de-icing agents

Sodium chloride is the dominant substance used in winter road maintenance and contains 40% sodium and 60% chloride (weight percent). In addition, it contains small amounts of calcium, magnesium and sulphate (Amundsen and Roseth 2007). Calcium chloride and Magnesium chloride are also used for de-icing, but less than sodium chloride. Both calcium chloride and Magnesium chloride are used for dust binding on gravel roads. Potassium chloride is used to a very small degree in winter road maintenance.

Chloride salts are water-soluble and dissociate easily when used on the road. Cations (Na^+, MG^{2+} Ca^{2+}) have different chemical, physical and biological properties and the effects of these in the soil and water will therefore be different. This will be discussed to a further extent in the sections below.

4.3 Organic based de-icing chemicals

Acetate (CH3COO-) and Formate (-COO) are easily degradable, water soluble and generally have low toxicity in water. Bioaccumulation is not expected. Both Acetate and Formate occur naturally in soil and the potential for adsorption is minimal due to the negative charge. COD/BOD for Acetate and Formate is respectively 1.07 / 0.7 and 0.35 / 0.27. Decay of formate in other words, is less oxygen demanding than acetate.

The most common de-icing chemicals that contain acetate are calcium magnesium acetate, Sodium acetate, Potassium acetate and sodium magnesium acetate, while the most common formate based de-icing chemicals are potassium formate and sodium formate. As for the chloride based de-icing chemicals an essential part of differences in environmental impact between these will be determined by characteristics of cations in the salt.

Propyleneglycol is readily biodegradable, has no potential for bioaccumulation, is completely soluble in water and has a very low toxicity. COD/BOD for the decay of propyleneglycol is 1.69/0.9 required, slightly higher than Acetate and Formate (Table 1).

Urea comprises amid-nitrogen $CO(NH_2)_2$ and is a white solid that is readily soluble in water (1080 g / l at 20° C). Urea is transported by water through the soil because it is an unloaded molecule. Urea has a very low toxicity, is bioaccumulative and easily broken down. The theoretical oxygen demand for the decay of urea is estimated to be 2.13 / 2.0 mg / l (Table 1).

There are various (bi) products from agriculture which are a group of organic substances and can be used for road deicing. These are different products of hydrated starch which all have low toxicity but which consume oxygen by decomposition in soil.

It is important to note that the COD/BOD values for de-icing chemicals (Table 1) are based on the complete decay of the solution with concentration 1 mg/*l* and that the chemicals can be used in other concentrations than this. The values in Table 1 are suitable therefore only as a comparison between chemicals.

5. The effects of deicing agents on soil

5.1 Effects of deicing agents containing chloride

Both inorganic (Cl_{inorg}) and organic (Cl_{inorg}) chloride occur in water, soil and air (Svensson et al. 2007). Cl_{inorg} are widely used as a tracer in hydrological studies because it is assumed that Cl_{inorg} is inert. Studies over the past 10 years show that Cl_{inorg} participates in a complex biogeochemical cycle where soil can act both as source and storage for Cl_{inorg} (Svensson et al. 2007). Examinations of rainfall, soil and runoff in a small catchment in Sweden showed, among other things that the soil is dominated by Cl_{org} and that the amount Cl_{inorg} going in and out of a catchments area is only approx. 3% of the total amount of chlorine (Cl_{org} + Cl_{inorg}) stored in the system. Runoff from the area was dominated by Cl_{inorg}. Drainage of Cl_{org} showed clear seasonal variations and Svensson et al. 2007 believe there is a need for further research on how chloride behaves in the soil and catchments in order to estimate the importance of road salt in such systems.

The effects of sodium chloride in the soil can be summarized in the following main groups (Green et al. 2008):

1. Effects on soil structure and changes in the composition of cations in the soil ion exchange sites and in soil solution. This can result in changes in pH in soil and soil solution, as well as changes in the biogeochemical cycle of pollutants in soil (for example, increased mobility and accessibility)
2. Increased potential for colloid transport in soils
3. Increased mobility of heavy metals in soil
4. Possible reduction in hydraulic conductivity in soil as pores get blocked by the particle and colloid transport

5.1.1 Effects on the nutrient balance in soil

Several studies show that the proportion of Na^+ bound in the soil along the road has increased with time and that proportion of Ca is reduced (for example Norrström and Jacks 1998; Norrström and Bergstedt 2001; Czerniawska-Kusza et al. 2004). This shows that Na^+ replacing Ca^{2+}, Mg^{2+}, Zn^{2+}, NH^{4+} and other cations from ion exchange complex and can cause the contents of nutrients to be reduce.

Leaching of Ca from the soil leads to lower base saturation (BS) in the soil, while the supply of air and water is reduced when the aggregate stability is reduced. This will reduce the growth conditions for plants (Norrström and Bergstedt 2001).

Road dust usually contains relatively large amounts of Ca and Mg so that the roadside surface soil often has a relatively high base saturation which decreases with increasing distance from the road (Bernhardt-Römermann et al.2006). This will however be an effect that occurs in the upper soil layer. Soil farther down in the profile where road dust isn't directly added will be more prone to leaching of Ca and Mg, with a subsequent reduction in BS, air supply and water supply.

5.1.2 Effects on soil structure

Soil that contains clay and silt are dependent on Ca (and to some extent Mg) to form stable aggregates. When concentration of Ca is too low the aggregates dissolve and clay particles and colloids in the soil become more exposed to the spread of water (dispersion of colloids). Another reason for the increased dispersion of colloids in the soil is that the Na-ion is surrounded by a layer of water which seems dispersed. Norrström and Jacks (1998) showed that leaching of Ca was higher near the road and that the leaching also occurred in the deeper layers of the soil (down to 0.5 m).

5.1.3 Improved colloidal transport in soil

Basically, increased salinity will stabilize colloids (less electrostatic repulsion), however due to leaching of Ca and Mg that are important for the stabilization of aggregates in the soil (Norrström 2005; Norrström and Bergstedt 2001; Backström et al. 2004), salt application will increase the colloidal carrier. This is especially true during precipitation episodes where salt is added to the soil in advance. In clay soil that contains many 2:1 clay-minerals (e.g. montmorillonite) the supply of sodium chloride could also lead to swelling and in some contexts, this development can result in an increased risk of quick clay.

The spread of colloids in soil is a function of how large the proportion of the cations on ion exchange sites in soil exists as Na (ESP) and ionic strength, or salinity in the soil solution. Norrström and Bergstedt (2000) refer to a classification system developed by Shainberg and Letey (1984) where the relationship SAR/ESP and conductivity of water was used to classify the risk of colloidal transport (SAR is an expression of the relationship between sodium and calcium + magnesium in the soil solution). According to this system, soil with conductivity lower than 0.2 mS/cm in the soil solution and relative SAR/ESP 0 to 3 will be exposed to colloidal transport. This applies, for example, to soil which over time has been exposed to a lot of sodium chloride and when rainfall added has a low ionic strength. Norrström and Bergstedt (2000) found that this classification applied fairly well with the transport of Pb in soil (column test), but that it was not always case. One of the explanations for this is that the system which Shainberg and Letey (1984) developed was based on other soil types with other pH values than those used in tests by Norrström and Bergstedt (2000).

pH is important for the stability of colloids in the soil because the charge of many soil colloids is pH-dependent and negatively charged colloids are believed to be more mobile than the positively-charged colloids. pH alters surface charge of clay minerals, Al-and Fe-oxides. Oxides are generally positively charged at pH values under 7 and negatively charged at pH values above 7. This will vary with the type of clay minerals and how crystalline the oxides are. pH of road dust is usually around 8 (Amundsen et al. 1999).

5.1.4 Mobilization of heavy metals

Temporary lowering of pH due to ion exchange processes (Bäckström et al. 2004; Appelo and Postma, 1996) will result in increased mobilization of metals. Investigations of soil water (taken with Prenart tension lysimeters) by two roads in Sweden showed that the mobilization of cadmium and zinc, primarily can be explained by acidification and the ion exchange processes. Mobilization of Pb were found to be very low, something Bäckström et al. (2004) explained with either a very good binding to the soil or that the mobile fraction of Pb in the soil had already leached out.

Generally increased leaching of colloids (both inorganic and organic) seems to result in increased mobility of Pb and Cu (Norrström 2005; Bäckström et al. 2004). The transport of colloids seems to happen after salting episodes with subsequent infusion of precipitation (or general water with low electrolyte levels).

Cadmium and zinc form chlorine- and hydroxide complexes and several have shown that salting leads to increased concentrations of these metals in the liquid phase in soil (Norrström 2005, Amrhein et al. 1992; Bauska and Goetz 1993).

Granato et al. (1995) reported increased concentrations of calcium, magnesium, potassium, manganese, barium, strontium, iron and zinc in groundwater downstream to roads where known quantities of sodium chloride, calcium magnesium acetate and calcium chloride were added. During the experiment the measurements of groundwater concentrations of a number of elements were taken as well as the pH levels in the groundwater both upstream and downstream of a motorway. Groundwater flow was perpendicular to the road which was salted and four different areas were monitored. The results showed different degrees of moderation throughout the year, from annual cycles with a variation between ca. 50 to 600 mg Cl / l, to more stable concentrations that ranged between 50-100 mg Cl / l. Although the added salt equaled more than 70 times the cation exchange capacity (CEC) in the area, the ion exchange reactions gave a noticeable change in water chemistry. It was also observed that a decrease in pH was equivalent to a doubling in the number of free H^+ ions. This happens even though there is an increase in pH in the unsaturated zone (in this case at ca.6m depth).

5.1.5 Reduced hydraulic conductivity in soil

Mobilization of colloids in the upper soil layer along the roadside may lead to reduced hydraulic conductivity if those colloids are released further down soil profile. If the texture of the soil is rough, as it is in most roadsides the colloids could be transported down to the groundwater. Reduced hydraulic conductivity will lead to reduced water transport and poor growth conditions for plants. Reduced air access for roots in soil will also be an effect of the aggregates collapsing and the colloids being transported through the soil profile (Environment Canada 2001).

5.2 Decomposition of organic de-icing chemicals

5.2.1 General

Complete decomposition (i.e. the decomposition to carbon dioxide and water) of organic compounds requires a certain amount of oxygen or other oxidizing agents (manganese and iron oxides, sulfate, nitrate) to be available. In the unsaturated zone more oxygen is available because the pores in the soil are not completely filled with water, while the air supply in the saturated zone is normally much lower. Because the diffusion of oxygen into the groundwater is slow, will oxygen depletion occur easily in this zone.

How roadside soils will act as a cleaning medium for organic compounds in the deicing agents depends on several factors other than oxygen access. Important conditions will be:

- Availability of nutrients (nitrogen, phosphorus)
- Temperature
- Grain size and flow conditions in the soil
- Retention in unsaturated zone above the groundwater level

Natural decomposition will take place in the vegetation, the soil and underlying sediments. Decomposition will largely take place in the spring after spring melt and ground frost thawing.

Decomposition capacity must be estimated taking into account the vegetation, nutrient status of soil, debris composition and profile thickness above the underlying groundwater.

To assess the fate of the de-icing chemicals there are three factors that are important to consider in relation to environmental impacts:

- Decomposition rates
- Oxygen consumption
- Transport properties (binding to soil surfaces or not)

Decomposition rates of organic deicing agents and oxygen consumption are commented on below; while we found that it would be more appropriate to comment on transport properties in conjunction with the chapter on groundwater (Chapter 10).

5.2.2 Decomposition rates

Organic de-icing chemicals such as K-Formate, Na-acetate, CaMg-acetate, Sodium-magnesium acetate, sodium formate, potassium formate and propylenglycol is water soluble and the organic part acts as carbon source for growth of fungi and bacteria in soil and will therefore decompose. Norwegian airports have been using some of these organic deicing agents for a long time, due to certain requirements in relation to the prevention of corrosion. A summary of the decomposition rates of the organic portion of the substances mentioned above are calculated from both field and laboratory experiments and presented in French et al., (2002), but are also shown here along with the results of a recent decomposition experiment (Table 2).

Table 2: The half-life for various deicing organic chemicals in soil.

Chemical component	Half-life, (days)	Location	Starting Concentration, (g / l)	Temperature, ° C	Reference
Propyleneglycol	15-45	Field	15-100	0-10	1)
		Laboratory			2)
Formate	18-34	Field	273	3-8(0)	3)
	7	Laboratory		20	4)
	0.5-4.5	Laboratory	0,50-1,50 (g formate C / l)	8-20	5)
Acetate	34	Field	105	0-10	1)
	4-18	Laboratory	0.5-5	20	1)
Acetate with N+P added	2.5	Laboratory	5	20	1)

1) French et al., 2002; 2) Linjordet 2007; 3) Calculated on the basis of data found in Hellstén et al. 2005a, K-Formate was added over a prolonged period, therefore a high and low value is presented, the lowest half-life is calculated on the based on the shortest retention time, 4) Converted from Roseth et al. 1998; 5) Oslo Airport, 2001.

The summary (Table 2) shows that decomposition increases with temperature, and that nutrient supplements (N + P) increase decomposition rates. It is therefore important to ensure enough retention in the unsaturated zone to ensure that the chemicals decay before melt water reaches the groundwater (French et al., 2001). Summer soil temperatures are often higher than otherwise and this is likely to increase decomposition rates. All the experiments referred to in Table 2, show an increase in manganese values from charged soil profiles. This can be best explained by manganese oxide being used as the electron acceptor for decomposition, this reduces manganese oxides to soluble $Mn^{2+.}$

In experiments by Hellstén et al. (2005b) potassium formate was added to a field lysimeter (1.7 m total depth) throughout a normal winter with snow cover in Finland. In addition to the discussion related to Table 2 above, the experiments showed an increased leaching of magnesium (Mg), zinc (Zn), barium (Ba), calcium (Ca), sodium (Na), and a higher pH. Total organic carbon (TOC) was still elevated at the end of the experiment, which indicated that the decomposition was incomplete. It further mentioned that the vegetation in the lysimeter did not tolerate the high charge that was used in the experiment. The vegetation here was representative of typical forest floor vegetation found in the area and contained, among other things moss, cranberry heath, and young birch (Hellstén et al. 2005b). It is also reported that the experiments with formate are still in progress in Finland along several stretches of road, and after 6 winter seasons formate has not yet been observed in groundwater samples (Salminen, pers.comm.).

In this context it may also be mentioned that the vegetation which received the highest doses of propyleneglycol and acetate in the experiments at Gardermoen were also burnt and brown.

5.2.3 Oxygen Consumption

In experiments with propyleneglycol and acetate added to a natural soil at Gardermoen, an increased concentration of dissolved iron and manganese was found (French et al., 2001), and further transport into the groundwater was observed (Øvstedal, pers.comm.). If the chemicals are broken down into the saturated zone then increased values of dissolved iron and manganese is likely and may cause disadvantages in relation to use for drinking water. In experiments with anaerobic decomposition of propyleneglycol Jaesche et al., (2006) concludes that one should ensure aerobic degradation of PG in order to avoid the accumulation of propionate (decomposition product), as well as mobilization of iron and manganese for further transport into the groundwater. In relation to the assessment of different organic deicing agents and urea, it is important to look at the figures for the theoretical oxygen consumption, reported in Table 1. These materials have low toxicity, are water soluble and have fairly similar transport properties, therefore can the lowest possible oxygen needs to be a good selection criterion. In the case of propyleneglycol can degradation under anaerobic conditions cause the formation of toxic mercaptan, which gives a rotten cabbage smell.

The relative difference in oxygen demand, measured as COD and BOD at various de-icing chemicals are shown in Table 1.

5.3 De-icing agents containing acetate

The degradation of acetate is strongly temperature dependent and at low temperatures can the calcium magnesium acetate concentration in the water get so high that the negative effects may occur as a result of O^2-deficiency (IHS and Gustafson 1996).

When it comes to effects of soil Fischel (2001) mentions the following factors:

Positive effects in soil

- Calcium and magnesium result in the stabilization of clay particles and thus increase the drainage and improved air access in the soil (generally a better soil structure).

- The advantage of CaMg-acetate rather than Na-acetate or K-acetate is that Ca and Mg binds more strongly to soil than Na and K and a lower leaching to groundwater.

Negative effects of soil

- High levels of calcium can temporarily lead to reduced availability of magnesium and potassium.
- CaMg-A can lead to increased mobilization of trace metals in the soil.
- Decomposition of acetate can lead to oxygen depletion in soil.
- Decomposition of acetate at temperatures above 10 º C occurs within 2 weeks, while at temperatures below 2 º C acetate is broken down slowly which increases the potential for leaching to groundwater.

Use of sodium and potassium acetate will not have the same positive effect on the stabilization of clay particles as the use of calcium magnesium acetate.

5.4 De-icing agents containing Formate

Rasa et al. 2006 demonstrated in experiments where the soil was incubated with potassium formate and sodium chloride that potassium formate led to a lower content of readily available (bioavailable) Cd in soil. One of the reasons is because the incubation with potassium formate resulted in reduced redox-potential of the soil and elevation of pH (4 to 7). Increases in pH will cause the binding of Cd to oxide surfaces to increase (more negative surfaces because of a greater degree of dissolved functional groups). In incubation experiments with potassium formate more than 80% of Cd bound in the soil was bound to oxides after incubation. Incubation with sodium chloride increased the proportion of readily available Cd due to the increased content of Cd-Cl complexes and ion exchange effects (Rasa et al. 2006). After incubation with sodium chloride was as much as 24% -39% of total Cd water soluble, most likely because of low binding capacity for various Cd-Cl complexes.

Hellstén and Nystén (2003) also found that organic deicing solutions based on acetate and formate leads to increased pH and alkalinity in the soil, compared to deicing solutions containing chloride. In column experiments that were carried out with 1m and 3.5m high pillars, was 70% of the added acetate and 82% of the added formate broken down in the 1m column (after 5 weeks), whilst the content of chloride in runoff was relatively equal the inlet water (no binding in the soil). After 5 months of treatment of the columns formate was only found in the column with sandy gravel that had no top soil with organic matter.

Column tests showed that more metal was extracted out by the use of sodium chloride than by the use of formate and acetate. The concentrations of Mn and Cd were, for example 6-7 times higher in the eluate from columns where sodium chloride was used rather than clean water. CaMg-acetate resulted in higher leaching of As, Pb, Ni and Zn compared with formate, whilst formate mobilized larger amounts of Cr and Mo than the acetate and chloride in these experiments. The contents of Mn and Na in runoff from the columns exceed drinking water criteria set for these parameters in Sweden (Hellstén and lately in 2003).

5.5 Propylenglycol

Propyleneglycol is broken down by other organic compounds such as lactic acid and Pyruvic acid under aerobic conditions. By-products of aerobic decomposition are considered to have low toxicity and break down easily. If propyleneglycol is broken down under anaerobic conditions, the by-products such n-propanol, propionate, acetate, mercaptan and methane are formed. Of these products mercaptan is the most unfortunate since this is a toxic gas that can cause odor disadvantages (smells like rotten cabbage).

5.6 Urea

Urea which is added to the soil will break down (hydrolysis) to ammonia and carbon dioxide under the influence of the soil enzyme urease. In the surface soil the urease activity is normally very high, and urea has a short lifespan. If conditions are dry then hydrolysis of urea will not occur.

Urea used for airport runway deicing during winter and for road deicing will hydrolyze much slower due to low temperatures and short-term contact with the surface soil which has a high urease activity. Swendsen (1997) proved both the hydrolysis of urea and subsequent nitrification of ammonia when urea was washed down through the sandy soil during snowmelt.

Use of urea provides large oxygen consumption in soil/recipients as a result of nitrification and it is organic deicing substances that consume the most oxygen by decomposition in the soil (Table 1). In recipients with high pH levels (above 7.5) ammonia can be formed after the hydrolysis of urea which is very toxic to fish and other aquatic organisms.

Many factors affect the turnover of urea in the soil, but the opportunities for optimization of process are mainly associated with liming, a well-developed and dense vegetation cover and phosphorus fertilization. Low pH levels (below 5.5) are able to prevent the effective nitrification of ammonium.

The best conditions for the decomposition of urea are obtained at pH 6-7, with a supply of P-fertilizer and an established close and permanent vegetation cover.

5.7 Carbohydrates

Studies from Sweden (Gustafsson and Gabrielsson 2006) on a mixture of a glucose/fructose solution in a salt solution show that on the basis of friction it is possible to replace 25% of the amount of salt with glucose/fructose. Similarly, it was shown that a mixture of 50% raw sugar and 50% sodium chloride had the same effect on friction than the use of 100% sodium chloride on the stretch of road that was tested. According to Gustafsson and Gabrielsson (2006) attempts should be followed up with new experiments and measurements.

Leaching experiments in the field and of soil packed in columns showed that the sugar solution (26.5 grams of sugar per liter) resulted in lower O_2 levels over time in eluate than sodium chloride (Thunqvist 2007). The total amount of lead and iron that was transported from the soil (column tests) was not significantly different from the columns where the salt (0.1 M sodium chloride) and sugar were used, but there were clear differences in the leaching pattern. The leaching of metals was greatest towards the end of the experiment in the columns irrigated with sodium chloride, while the leaching was more evenly distributed over the trial period by irrigation with sugar solution. Organic material binds well to lead and iron – copper is also leached out in the same pattern as these metals irrigated with the sugar solution. Increased leaching of lead and iron towards the end of the experiment (distilled water in the last 4 days) due to sodium replacing calcium and magnesium from ion exchange sites in the soil, leading to reduced stability of colloids and increased dispersion and thus increased transport when ionic strength is reduced (distilled water).

Leaching of cadmium and zinc is higher with the use of sodium chloride than by the use of sugar solution.

This is because as mentioned, these metals form soluble chloride complexes in soils (Amrhein et al. 1992; Bauska and Goetz 1993). A possible negative effect of the use of sugar is that sugar on the road or accumulated on roadside vegetation, can attract animals. It has been shown that salt also has this effect. Experiments carried out in Sweden where salt and sugar stones were positioned along the road showed that there was a tendency for both moose and cow licked more from the sugar stones than the salt stones (Hedin 2006). The tests, however, says nothing about whether the placement of sugar stones increases the number of animals along the road.

Examples of other types of carbohydrates that are used for deicing and road maintenance are various organic by-products (the liquid) from agriculture (such as Ice Ban, 50% Ice Ban + 50% $MgCl_2$, http://www.meltsnow.com/pdf/iceban.pdf). It is likely that such products will break down in the soil and possibly lead to the mobilization of iron and manganese, and possibly other metals (see above).

Generally, the soil's capacity to break down organic de-icing chemicals determines the extent to which the organic substances will be able to reach surface water and possibly lead to oxygen depletion.

5.8 Critical limits for de-icing chemicals in soil

In order to determine critical limits for soil chemical parameters, knowledge of the relationship between chemical and physical parameters in soil and the biological response is required. Currently there is not enough knowledge to determine the critical limits of the chemical or physical composition of soil (single parameters or combinations of parameters).

The application of salt is seasonally dependant and simple changes that occur in soil in connection with the application in winter and spring will be reversible. This entails, for example, that the degradation capacity of organic based deicing agents in soil may increase through the summer and autumn so that the potential for degradation during the next salt application (next winter) has increased in relation to the end of the previous salting season. How the decomposition rates possibly change over time may be investigated further.

Whether or to what extent the soil "regenerates" after the application of deicing agents containing chloride (especially sodium chloride) are also uncertain. Such knowledge is necessary in order to determine critical loads for soil.

For soil, it is more relevant to establish sensitivity criteria or hazard indices instead of critical limits. In such contexts, it is necessary to understand the relationship between roadside soil types, their properties (for example, clay content) and any changes in composition over time. An example of this is the salt hazard index (SHI) (Environment Canada 1999) defined as;

SHI = ESR (Exchangeable sodium ratio) x clay content in soil (%)

Soil receiving salt periodically, and which has high clay content will have a high SHI. As mentioned above, these areas will be particularly vulnerable to effects such as increased colloidal transport to the groundwater, reduced hydraulic conductivity and reduced air access, and also a possible increase in erosion as a result of poorer aggregate stability in the soil. By measuring the ESR over time and determining the clay content in soil on Norwegian roadsides, can such an index (or equivalent index) also be useful in Norway. A similar useful indicator which can be used with ESR is to measure the relationship between Na and Ca + Mg in soil and water whilst simultaneously measuring soil conductivity (see above).

Critical loads for strong acid in forest soils are set to ensure that the relationship between base cations and aluminum ions in the soil water will be greater than 1 (Larssen and Høgåsen 2003). When the leaching of Ca and Mg is an effect of road salting, this relationship can also be used for areas where sodium chloride is applied. More detailed knowledge of the relationship between ESR and the effects on flora and fauna is needed to develop a SHI from sensitivity index to a critical limit.

5.9 Discussion and conclusions

The effect of sodium chloride is greatest closest to the roadside where the concentration of Na in soil is shown to increase over time. This leads to reduced hydraulic conductivity, reduced permeability and therefore reduced access to water and air for plants that live along the road. Use of calcium chloride, magnesium chloride and calcium magnesium acetate is more favorable in order to avoid reduced hydraulic conductivity capacity in the soil and increase colloidal transport. Based on this, a mixture of sodium chloride and $CaCl_2$ or $MgCl_2$ for example would also be more beneficial for maintaining both the nutritional balance in the soil and the physical condition.

Both chloride-based and organic-based deicing agents have been shown to lead to increase the mobility of heavy metals in soil. For chloride-based deicing agents, this is due to both the increased colloidal transport as a result of reduced aggregate stability and the increased quantity of dissolved chloride complexes (especially important for Cd and Zn), while the effect of the organic deicing agents is primarily due to increased colloidal transport because of the reduction in Fe-and Mn-oxides and, plus that some organic compounds can bind heavy metals and with that increase mobility. In many cases a decrease in heavy metal mobility has also been noted due to the increased pH during the decomposition of organic deicing agents.

Measurements of soil chemistry (total and soluble content in the soil) should be performed to a greater extent over time in several locations because this allows distinguishing between short- and long-term effects of salting (salt spray vs. changing growth conditions in soil). Decomposition of organic materials (both Acetate and Formate) in different soil types should be part of such experiments.

6. Effects of de-icing chemicals on roadside vegetation

6.1 Introduction

Application of large amounts of sodium chloride causes damage to plants. This is well known in coastal areas, but many years ago damage to trees along roads in connection with winter salting was recorded in Scandinavia (Hedvard 1972, Sanda 1973, 1976). Furthermore, there are numerous reports from other parts of Europe and North America. A number of these are reviewed by Horntvedt (1975). Salt damage to trees along the roads had already been observed in Norway more than 50 years ago, but this damage was due to calcium chloride (Traaen 1958). In North America, damage caused by winter salting with sodium chloride is reported back to 1950 - and-60's (Lacasse and Rich 1964, Baker 1965).

The damage along the roads has been variable both in location and time and between species. It is perhaps not so strange that the significance of the damage has been somewhat controversial. The effect of salt along roads is described by several authors and especially the trees along the city streets show clear inflicted injuries.

In the urban environment in Norway road salting seems to be a major cause of poor well-being and reduced growth in city trees (Sanda 1973, P.A. Pedersen 1984, 1990, Fostad and P.A. Pedersen 1997a). Damage has also been proven in natural areas, both internationally and in Norway. Over the past 20 years, there has been an increase in the number of roads salted and amount of salt applied per distance. Norwegian Public Roads Administration therefore funded a multi-year interdisciplinary project with surveys of the environmental effects of road salting, both in field studies and cultivated trials under controlled conditions (P.A. Pedersen and Fostad 1996, Rohr 1996, Åstebøl and Soldal 1996). A review of relevant literature on the effects on vegetation (P.A. Pedersen and Gjems 1996) was also undertaken. A brief presentation of the results of the project is given in Åstebøl et al. (1996). In wake of the project the surveillance of a small wooded area with major damage was completed (P.A. Pedersen et al. 2002).

The literature review and results from the Norwegian studies mentioned above add considerable weight to the literature compilation presented here. In many contexts it is shown in older literature. Older literature provides a thorough and careful description of the action mechanisms of plants and thus covers the issues we consider relevant to literature compilation well.

6.2 Effects on plants

6.2.1 Symptoms

The symptoms vary with species and whether the salt is applied directly to the above ground parts of plants or taken up via the root system. Symptoms described by many authors (Holmes 1961, Sauer 1967, Walton 1969, Sanda 1973, Hofstra *et al.* 1979, Menlove 1973, Bäckman and Folkeson 1995, PA Pedersen and Fostad 1996, Randrup and Pedersen LB 1996, LB Pedersen and Inger Slev 2007), and can be summarized for trees as:

Direct salt spray gives discoloration of needles and leaves, often only on one side of the tree. Eventually necrosis (dead tissue) develops in the discolored parts. Necrosis often develops on the leaf edges, but this can also be quite irregular. Brown, necrotic needles eventually fall off. Coniferous and winter green species that have developed a leaf mass during the salting season are particularly affected by road salting, but deciduous species can also be damaged significantly. Both the buds and the vascular cambium can be damaged or killed with delayed defoliation or dead branches as a result. Direct salt spray from the road generally causes solitary damage to larger individuals in both the coniferous and deciduous species, while small individuals can be killed completely. In Norway, quite extensive damage along some stretches of road caused by salt spray has been recorded over the last few years in the eastern part of southern Norway (PA Pedersen 2007).

Salt uptake from the soil results in discoloration (chlorosis) of the leaf edge which gradually becomes necrotic. Necrosis follows the leaf edge and increases in width or can cover the entire leaf when strongly exposed. In conifers needles colour brown from the tip towards the base and defoliate later. Strong damage may cause all the mature needles to be killed, whilst shoots are green. Salt uptake through the roots rarely causes the typical one-sided damage to the canopy caused by salt spray, but can in some cases follow a spiral pattern or frame some branches particularly strong probably due to differences in exposure of the individual main roots (referred to in PA Pedersen and Fostad 1996). Such damage usually worsens during the summer, especially in dry conditions. In the case of prolonged or particularly strong effects branches of increasing size die and, at worst, the entire tree.

6.2.2 Primary and secondary effects of high salt concentrations

The effects of sodium chloride on plants depend on the type of exposure, salt concentration and exposure time. Levitt (1980) groups the action mechanisms as follows:

a. Primary direct salt damage

Direct salt damage caused by sudden increases in concentration which result in lasting damage to cell membranes. Direct damage is difficult to distinguish from indirect damage.

b. The primary indirect salt damage

Reduced growth and development occurs as a result of metabolic disturbances in photosynthesis, respiration, protein metabolism, nucleic acid formation, enzyme activity and the formation of toxic by-products.

Secondary salt stress can be divided into:

a. Osmotic stress

High salt content in soil can lead to reduced water absorption in plants because the osmotic potential is too low in relation to plant tissue, so-called physiological drought.

b. Ionic competition

Nutritional deficiency can occur or is enhanced as a result of competition between the ions in the dissolved sodium chloride and nutrients in the soil.

The effect of salt on plants depends on the salt composition and concentration. Sodium chloride has little toxicity in small and moderate concentrations, but damages many plant species in high concentrations. Levitt (1980) therefore distinguishes between salt stress and ionic stress. Ionic stress occurs even at low concentrations of salts that contain toxic ions, while salt stress occurs as an osmotic interference when the water potential is lowered significantly in the plants. The effects of sodium chloride are therefore related to salt stress. Salt stress can be of primary and secondary grades. Injuries resulting from osmotic disturbances in the plant tissue are considered primary, while problems with water absorption due to low water potential in soil and any possible changes in soil structure due to exchange of calcium with sodium are of secondary grade. Addition of sodium chloride also causes increased pH (cation exchange reactions) which may affect the solubility and hence the availability of plant nutrients.

Osmotic disturbance occurs at very high salt concentrations in the soil or after accumulation in the foliage over time. Lower salt concentrations can also be disruptive to plant growth because the ions are competing with the necessary nutrients for uptake in plant roots (Schachtshabel *et al.* 1982).

6.3 Uptake and transport of NaCl in plants

Transport of sodium and chloride in plants varies between plant species. Both elements are taken up easily by roots, but further transport is different. In particular, the salt-tolerant plants (halophytes) transport a lot of sodium through the leaves whilst this transport is low in salt sensitive plants (Poljakoff-Mayber and Gale 1975). In the latter group of plants sodium often retained in the roots or the stem/stalk. The relationship between chloride and sodium varies between species that are considered salt sensitive.

Fostad and P.A. Pedersen (2000) found higher Cl/Na conditions in the Norway maple than in spruce. In the sugar maple the content of chloride is from 8 to 250 times higher than that of sodium in the leaves, depending on the level of exposure (Baker 1965).

Leaf samples from deciduous trees under different salt-exposure typically show low stable sodium content, while the chloride content varies to a greater extent (P.A. Pedersen and Fostad 1996). There are a number of studies that document that salt damage is primarily caused by the accumulation of chloride in the foliage. Many of these are cited in Levitt (1980). Therefore the chloride content in the leaves correspond more with the extent of leaf damage that the sodium content (Baker 1965, Walton 1969, Lumis *et al.* 1976). Fuhrer and Erismann (1980) found that the area of necrotic tissue on leaves of the horse chestnut was directly proportional to the chloride content in the range of 1-1.6% of dry weight. They found also that chloride content increased approximately linearly with the leaves' age up to approx. 100 days and then levels off. The importance of exposure time, is also emphasized by Fostad and P.A. Pedersen (2000) as cultivated experiments with spruce found far less damage in November, 4 months after the transfer of salt, than in July the following year. Rains (1972) produced an extensive literature overview of salt transport in plants.

Bogemans *et al.* (1988) showed that sodium concentrations in the branches of spruce which were exposed to sodium chloride decreased when calcium was added. They also proved reduced chloride uptake at high chloride doses if calcium was added simultaneously. Good calcium access provided improved K/Na ratio which is considered positive for plant health. However they point out that large chloride uptake confiscates a lot of K+ in order to balance Cl- in translocation which can cause indirect potassium deficiency.

Ionic competition is described in more detail in Salisbury and Ross (1992). Potassium deficiency may be a problem for plants exposed to high concentrations of sodium. The concentrations of calcium are, however, crucial for plants' ability to take up potassium competing with sodium. If calcium is applied to the plant prior to sodium, it is shown that the negative consequences of sodium addition can be eliminated. It is believed that this is because calcium protects cell membrane against sodium and the Ca/Na- relationship in the root system.

6.4 Physiological mechanisms

A detailed description of the physiological mechanisms that trigger damage at high levels of sodium chloride is beyond the objective of this literature compilation. The effects of high salt concentrations are, however, presented and discussed in Strogonov (1973) and Jennings (1976). Strogonov (1973) summarizes the mechanisms as follows: Reduction of the osmotic potential of cells results in dehydration of the protoplasm. This affects enzymes and metabolism and leads to the formation of toxic compounds. Toxic compounds are formed largely as a result of disturbances in nitrogen turnover, which in turn leads to the accumulation of among other things ammonia, certain amino acids, pigments and sulfur compounds, on the other hand, by-products that have protective ability are also formed.

Fedina *et al.* (1994) found that sodium chloride exposure found increased production of the osmotic active proline in pea plants. They also found that infusion of the hormone abscisic acid (ABA) eliminated the inhibitory effect of sodium chloride on photosynthesis. They therefore consider the ability to form an increased amount of hormone ABA as an adaptation to salt stress in plants.

Kayama *et al.* (2003) examined a number of physiological parameters in two species of spruce *(Picea abies* and *P.glehni)* that were planted along a salted road and found a reduction in transpiration, photosynthesis and reduced ectomycorrhiza - infection of the roots.

6.5 Adaptation to high salt concentrations

In many marine (near seawater) and arid (water-poor) environments are the levels of sodium chloride or other salts generally high. In Salisbury and Ross (1992) the various mechanisms to survive at high salt concentrations are described: *Salt accumulating plants* take up large amounts of salt, but can withstand this. Usually the salt ions are stored in the vacuole (the bladder inside the cell). *Salt-regulating plants* have the ability to exclude salt, so that root uptake is reduced and the concentration in the plant is kept moderate. Osmotic balance is maintained through the production of organic compounds, for example, proline. *Succulent plants* dilute the salt concentration whilst simultaneously taking up a lot of water. *Fast-growing plants* have the ability to keep salt concentration low by constantly forming new cells. Older parts of plants that have accumulated much a lot of salt can eventually get rid of salt by gradually losing leaves. *Salt eliminating plants* reduces salinity by transporting the salt into the surface of the plant where it remains as a coating which is washed off in rain or gathered in special sacs. At the biochemical level adjustment can occur by the breakdown of harmful substances formed during salt stress, or by having particularly robust enzyme systems.

Since the adjustment mechanisms are different, there will not always be good correlation between extent of damage and content of for example, chloride in the plant. In various species of shrubs along the Norwegian roads damage occurs at different chloride content, even within the same species (PA Pedersen and Fostad 1996). This illustrates that the plants have different ability to defuse or prevent the harmful effects of salt ions after uptake.

Adaptation mechanisms have limitations in relation to exposure level. Levitt (1980) refers to investigations where it is demonstrated that plant roots can only exclude salt ions up to a certain concentration. When this concentration is exceeded the uptake increases dramatically and damage occurs.

6.6 Significance of changes in soil

Effects of soil are examined more thoroughly in another chapter, here are some key points of relevance to plants. Addition of large amounts of sodium chloride to the soil affects the ionic composition and will cause the calcium and magnesium that is bound to the soil to be released to a certain extent and replaced with sodium. This is particularly unfortunate in the clay soils which obtain poorer structure and become sludge when calcium is replaced (Schachschabel *et al.* 1982). In general, this process will contribute to plant nutrients is leaching down through the soil profile and out of reach for the roots. A Norwegian survey demonstrates a reduced pH in soil water on roadsides (Røhr 1996).

Changes in soil chemical conditions caused by sodium chloride as a result of sea salt spray have also previously been found in Denmark, where changes in aluminum concentrations and a decrease in calcium/aluminum ratio and pH in salt-exposed areas (LB Pedersen 1993) was found. In this study the decrease in pH is explained as the exchanged aluminum hydrolyses so that protons are released. On Rv2 in the county of Hedmark in Norway, relatively high concentrations of free aluminum in the soil solution have been found (approximately 2.8 mg/l (Røhr 1996).

Concentrations of aluminum at this level result in growth changes in Norway spruce (Göransson & Eldhuset 1991). Along the mentioned road stretch calcium concentrations of approximately 100-140 mg/l have been recorded (Røhr 1996). This gives a Ca/Al ratio of ca. 10-30 which is far above that which Eldhuset (1988) refers to as minimum requirements for normal root development of trees. There is therefore no reason to believe that the release of aluminum as a result of road salt has caused damage to the forest in this area. Effects of sodium chloride on conditions in the soil are also described further in a literature compilation by Røhr (1995).

6.7 Testing of salt tolerance in plants

6.7.1 Variation in salt tolerance between species

In many surveys are different species tested with a variety of salt exposure, and there are several literature studies which rank species in respect to salt tolerance.

Such rankings have large sources of error because the species have been tested under different conditions, data from both field and laboratory investigations are used and it might not have been considered as to whether exposure occurred via air or soil. Furthermore, the variation within each species is rarely taken into account. Based on literature (both field records and the control of salt supply) Sanda (1973) made lists of ranking of the species salt tolerance. Later new lists were created (Dobson 1991, Barker *et al.* 2003.). Randrup and L.B. Pedersen (1996) present a comprehensive list with reference to a number of species and also try to make a conclusion for each individual species. They, however, emphasize the significant sources of error in the ranking. Below individual studies are referred to in which testing is conducted under controlled conditions.

Based on laboratory experiments Townsend and Kwolek (1987) found that of 13 pine species *Pinus thunbergii, Pinus ponderosa* and *Pinus nigra* were most tolerant to sodium chloride-spray. Least tolerant to salt spray were: *P. strobus, P. banksiana, P. cembra, P. peuce* and *P. densiflora.* The species *P. strobiformis, P. aristata, P. parviflora, P. resinosa* and *P. silvestris* were intermediate. In an experiment conducted by Fostad and Pedersen (2000) the pine was by far the most tolerant of the four species, spruce, birch, pine and maple tip. Spruce was particularly sensitive. They also showed that the extent of damage varied greatly with soil type.

Townsend (1984) examined the tolerance of 6 species of seedlings and found that the most tolerant species were *Ginkgo biloba (Ginkgo), Gleditsia triacanthos* (honey locust) and *Sophora japonica* (pagoda tree), while the most sensitive species to sodium chloride were *Platanus occidentalis* (American sycamore) and *Cornus florida (flowering dogwood). Pinus strobus* (eastern white pine) was moderately tolerant to salt in relation to the other species in this experiment. Paludan-Muller *et al.* (2002) injected sodium chloride to four species of deciduous trees both via the soil and directly on the buds and bark (to simulate salt spray). They found that direct salt exposure on the trees led to delayed bud break and that the Horse Chestnut was the most tolerant to salt spray and Beech were the most sensitive. Small leaved linden, however, demonstrated the most salt uptake from the soil.

Thompson and Rutter (1986) treated 11 species of shrubs with different concentrations of sodium chloride as direct spray or added to the soil. The most tolerant species was *Hippophaë rhamnoides* and the most sensitive was *Crataegus monogyna.* They found no clear relationship between tolerance for sodium chloride applied as spray and to the soil.

Salt tolerances in grasses have been studied by Sanda (1978). He examined 29 different grass varieties and ranked them in relation to salt tolerance. The most tolerant grass 'bent grass' cultivar was *Festuca rubra* '18 DP 'and the least tolerant cultivar was *Agrostis canina* . It turned out that the varieties of *Festuca rubra* (red fescue) was generally salt tolerant. Sanda found that there was no direct correlation of salt tolerance between grass varieties during germination and full grown plants. The species *Lolium perenne* (perennial ryegrass) proved to be the most tolerant during germination and was also the species that grew fastest. Aamlid and Hanslin (2006) points out that grass is the most salt sensitive in the stub-and start-up phase and based on the cultivation attempts rank the species used in green areas as: perennial ryegrass (most tolerant)> red fescue> creeping bentgrass> hard fescue> kentucky bluegrass > sheeps fescue > common bent (least tolerant). They also remind that there are differences within species.

Effects of sodium chloride on perennials are weakly documented. Søyland (2006) proved, however, significant differences in salt tolerance between species used in the green areas along the roads. For example, *macrorrhizum Geranium* (Geranium blue splish splash), which is widely used as ground cover, had relatively good salt tolerance while *Geranium magnificum* was less tolerant and use was discouraged even at moderate sodium chloride exposure.

6.7.2 Genetic variation within species

As proven by Sanda (1978) maritime ecotypes may have greater salt tolerance than the continental ecotypes. However, variation can often be detected, although it is unclear whether this is due to a prior selection. Dochinger and Townsend (1979) studied the response to ozone and sodium chloride of three offspring groups of red maple *(Acer rubrum)* and concluded that genetic differences and environmental factors were crucial for the response.

In some studies, widely varying content of sodium and chloride is found in leaves of different individuals of the same species under apparently equal conditions (Baker 1965). This indicates that there is considerable genetic variation in the ability to take up and relocate salt. He found that leaf damage was correlated with chloride content, but not with sodium content. He also found examples of high sodium values in the undamaged trees and low values in injured trees. This suggests that the ability to take up sodium varies independently of the ability to take up chloride. Selecting the source of seed may therefore be important for tree species that are considered salt sensitive. Fostad and P.A. Pedersen (2000) found clear differences in the half sibling families of Norway maple *(Acer platanoides)* and common spruce *(Picea abies).*

6.8 Effects as a result of a natural salt influence

The vegetation along the coast, recieved large quantities of sodium chloride by spray caused by strong inlands wind. Therefore, burn damage occurs regularly at varying distance from the sea along the west coast of Norway. Since the most powerful storms usually occur in winter months it is winter green vegetation, especially conifers, which suffer the greatest damage.

After the hurricane winter of 1992 obvious damage to the spruce on the east side of the Trondheim Fjord arose, hundreds of meters from the fjord (P.A. Pedersen 1993). In the spring of 1990, significant salt damage to the spruce was found. Major damage was found both in the very outer regions and in the fjord arms (e.g. at Jølster). Damage was correlated with chloride content in the needles (Aamlid 1995). Although it is usually conifers that are affected the most, after the storms in this part of the season, deciduous trees could also be damaged. Even in less exposed parts of the coast, salt damage caused by salt spray has been recorded. In August 1988, a number of species of trees and shrubs were damaged in Hvaler in Østfold, and increased concentrations of sodium and chloride were found in birch leaves up to 30km from the sea (Horntvedt and Aamlid 1989).

The amount of salt that is transported in a storm varies greatly with wind speed. According to Boyce (1954) increase in wind speed from 7 to ca. 13m/s gives almost 6 times as much salt deposition on the vegetation.

In Danish spruce population with large salt damage approximately 10 km from the coast, sodium chloride values up to 2.2 meq/kg soil are found, i.e. 129 mg/kg soil (LB Pedersen 1993). These values are clearly lower than those found near the Norwegian roads (P.A. Pedersen and Fostad 1996).

6.9 Damage to vegetation along salted roads

There are many different types of studies of vegetation damage caused by road salting, both in urban environments and in the forest. Special episodes with locally extensive damage have often been the basis for investigations and the results therefore provide very location and climate dependent information. In some cases the focus is on the effects of salt spray, while uptake through the roots is stressed in other cases.

6.9.1 Spray damage

Traffic swirls up the salty water from the road. This water is spread as aerosols over longer or shorter distances depending on the speed limit, traffic density and climatic conditions. Lumis *et al.* (1976) found in his investigation that the salt content in the twigs along salted roads varied with temperature and precipitation during the sampling period. McBean and Al-Nassri (1987) found that the maximum dispersal distance for salt was proportional to driving speed. In the Norwegian studies along the roads outside densely populated areas, spray damage is observed more frequently at 5-8 m from the road, and occasionally at 15 m or more (P.A. Pedersen and Fostad 1996). The damage was also generally limited to approximately 2-4 m above the road. This is in accordance with Bäckman (1980), who mainly found damage up to 10 m. It is shown in multiple studies that the dispersal of air-transported sodium chloride decreases exponentially with increasing distance from the road (Blomqvist and Johansson 1999). Investigations of salt deposits at different distances from the Norwegian roads show that most of the salt dispersal occurs within approximately 7m from the road. However, there seems only to be a small portion of the salt that is spread through the air, probably ca. 10-25% (P.A. Pedersen and Fostad 1996). Blomqvist and Johansson (1999) concluded, however, that 20-63% of the applied salt was transported through the air and deposited from 2 to 40 m from the road. Dispersal of air-transported sodium is also described by L.B. Pedersen *et al.* (2000) and also describes measures to protect vegetation. They found a small but measurable effect of protection by means of straw mats. In a multi-year study, L.B. Pedersen and Krag (2005) found a significant reduction in the content of sodium chloride in the upper 25 cm soil layers than in the areas which were protected by straw mats (70-90 cm high) after the salting season, compared with unprotected areas.

The dispersal of salt by spray is affected by vegetation at the site. Bäckman (1980) found that salinity in soil decreased faster with increasing distance from the road in dense forest than in thinly forested areas. It is therefore pointed out that a forest near the road provides protection against salt spray. The vegetation's ability to capture salt spray is probably the reason Thompson *et al.* (1986) found 50% more sodium in the soil under shrubs than in open ground on a central reservation. After the winter of 1993/94 unusually extensive salt damage to vegetation along the E20 and Rv 48 in Skaraborg in southern Sweden was recorded. Here spray damage was found at least 50 m from the road. Such damage occurred only in areas where protective vegetation was not found close to the road (Bäckman and Folkeson 1995). They found 1.7% and 2.0% sodium chloride in the dying pine needles. This is more than double the values P.A. Pedersen and Fostad (1996) recorded along a Norwegian road. One explanation of this can be that much of the salt is outside the needles and therefore does not reflect actual tolerance limits. This is also the probable reason for the high sodium content found in the needles. This is also probably the explanation for the content of sodium in the needles is so high. Mature needles of conifers which have been exposed to salt spray have a higher Na/Cl ratio than the previous year’s needles (Bäckman and Folkeson 1995, P.A. Pedersen and Fostad 1996, Viskari and Kärenlampi 1999). In a more recent survey in east part of Norway there are proven damages found at a greater distance than previously. Significant spray damage on birch found 70m from the road were found at the speed limit 80-100km/h, but other species were also damaged at a relatively large distance from the road. Large local differences in the extent of damage were observed to be greatest in open areas and are expected to depend on local wind conditions (Pedersen 2007). Generally birch is very susceptible to damage, but also hazel, Gray Alder, black alder, beech, common hornbeam and common pine seemed to be very delicate. White poplar, Norway maple, elm and ash were a little sensitive. First the buds were damaged, then the xylem and young phloem.

Kelsey and Hootman (1992) argue in a U.S. study that salt spray along main roads reach a height of at least 15m up to approximately 70m from the road. They found the salt spray damage on eastern white pine *(Pinus strobus)* at a distance of 277m from the road and supposedly even coniferous damage on pine 378m from the road due salt spray from the road. Sodium concentrations in the needles, however, were quite low, ca. 1000ppm (approx. 0.1%). This study was conducted in an area that was affected by an extensive road system with parallel paths and many lanes. Such high exposure is unlikely in Norway. The extent of damages i.e. a height of 15m and up to 70m is however, of the same order of that registered in Norway.

Bäckman and Folkeson (1995) observed damage to a number of species, both coniferous and deciduous. Horse Chestnut *(Aesculus hippocastanum)* seems to be fairly tolerant. In this study high sodium concentration in the mature needles were recorded, up to approximately 8m above the road and then strongly decreasing concentrations up to ca. 18 m. Chloride concentrations declined considerably weaker, which is not surprising since the uptake of chloride from the soil leads to high levels even at higher altitudes. Bäckman and Folkeson (1995) assumed that dispersal of salt at a late stage was a major cause of the unusually large damages seen in Sweden.

Damage caused by salt spray depends on the plant size. Large trees often only get damaged on the lower parts of the trunk, whilst young plants are killed. Spray damage is therefore a problem for the hedges along the road, especially the winter greens. A hedge should be tight and fresh even at the bottom, but this can be difficult to achieve near salted roads. Lumis *et al.* (1973) registered the salt spray damage to 75 species that were located 8-40m from roads in Canada. Norway maple *(Acer platanoides)* and Horse Chestnut *(Aesculus hippocastanum)* were among the undamaged deciduous species, while birch species displayed significant signs of damage. Colorado blue spruce *(Picea pungens)* and the mountain pine *(Pinus mugo)* had the least damage among the conifers.

6.9.2 Damage caused by high salt content in the soil

Salt damage to trees in urban areas is generally caused by uptake through the roots, and there have been a number of surveys in urban centers used to map tolerance differences between different species (Hedvard 1972, Sanda 1976, Liebert 1978, Simini and Leone 1986, Leh 1990, 1992, Fostad and P.A.Pedersen 1997b).

Along the roads in natural areas, fewer surveys have been completed. In Sweden, obvious injuries on spruce were observed in the late 70th century (Bäckman 1980). Although the main cause of this damage was thought to be salt spray, in some cases the concentrations in the soil were so high (up to approx. 400 ppm Na and approx. 700 ppm Cl) that the damage due to uptake through the roots can not be ruled out.

In Norway, throughout several years significant damage to the forest near the E6 and Rv2 in Hedmark has been observed (P.A. Pedersen and Fostad 1996). In the study it was concluded among other things the following:

- damage had occurred in particular locations where drainage was poor and the terrain was lower than the road

- common spruce was damaged the most and in several cases bark beetle attacks took place after salt damage had occurred

- birch seemed to be fairly tolerant of high salt content in soil, but was sensitive to the salt spray

- blueberries had a high uptake of chloride and were strongly damaged

- common pine was not injured as a result of uptake through the roots, but was sensitive to the spray

- salinity in the upper soil layer had a tendency to increase within a certain distance from the road

- the damage was more extensive in Østlandet (Norway) than in Rogaland (Norway)

- in Østlandet (Norway) the salt content in the soil was higher in the autumn than in the spring/early summer, whilst the opposite was the case in Rogaland (Norway)

In the aforementioned study, highly variable levels of salt in the soil were found depending on the soil type and year. In more detailed investigations by Røhr (1996) it was confirmed that the soil in areas with particularly intense damage had a dense layer that provided opportunities for the accumulation of temporary groundwater in or near the trees root zone. This temporary groundwater was clearly affected by salt (see also Røhr 1995).

He found examples of large variation in salt concentrations down through the soil profile. Because of such conditions damage was found in some places more than 50m from the road (P.A. Pedersen and Fostad 1996). In supplementary investigations done at a particular locality with major damage (by the E6 Norvimarka in Stange, Norway) later, damage to spruce was found up to a distance of 100m from the road and high chloride concentrations in the upper soil layer at a large distance from the road. Temporary groundwater at the site had higher concentrations of sodium chloride (P.A. Pedersen *et al.* 2002).

In the U.S. extreme cases have been reported where in a day the saline ground water has flowed out to a distance of 200-300m from the road and damaged the forests (Jordan 1986).

Bäckman and Folkeson (1995) found sodium and chloride concentrations in the soil up to 8m from the road reached up to 800-1000 and 900 ppm which is comparable with the most deprived areas in Norway. They found, like Røhr (1996) changes in salt levels down through the soil profile. The levels could be both increasing and decreasing.

Variation in local soil and climate conditions is crucial for the extent of salt damage to vegetation. For example, Bäckman (1980) found little correlation between the extent of damage, salt levels in soil and dispersed quantity. This shows that the use of randomized sampling to map salt stress may provide a distorted picture of situation. The samples should be taken and the analytical results evaluated on the basis of local conditions. Local conditions are undoubtedly crucial to how the concentration in soil varies throughout the year. In a British study Thompson *et al.* (1986) discovered that the highest salinity values were found in roadside soil in March-May and clearly decreased in value throughout the autumn. However, at several Norwegian localities high and partly increasing values were found in the autumn (P.A. Pedersen and Fostad 1996, Rohr 1996). In such cases vegetation is exposed to continuous high values and the probability of damage is great.

In urban environments, salt transportation through the soil depends on the conditions of the road technical. L.B. Pedersen and Holgersen (2006) claim that in one case the elevation of the central divide gave close to an 80% reduction in salt concentration in the soil solution.

Salt damage along the roads as a result of uptake from the soil varies greatly between different species. For example, oak is considered as a relatively tolerant tree species to high salt content in soil (Shaw and Hodson 1981). Zolg and Bornkamm (1983) found no accumulation of sodium and chloride in the leaves of oak *(Quercus robur)* on the roadside, whilst Horse Chestnut *(Aesculus hippocastanum)* and crime linden *(Tilia euchlora)* had significant uptake of chloride. Trockner and Albert (1986a) points out that salt tolerance depend on a tree's ability to retain salt in the root and stem. They found that species of birch and oak had little salt accumulation in the roots and leaves and argues that species from hot and dry climates can withstand salt stress the best.

Urban trees can achieve very high chloride concentrations. In heavily damaged leaves of Horse Chestnut and beech Dragsted (1977) found that the chloride content was respectively 2.23 and 1.75% in leaf dry matter.

6.9.3 Changes in the local species composition

The vegetation in coastal areas is adapted to both wind and sea spray over a long time period. This has led to a particular species composition, with particular salt-tolerant species in the coastal areas, but there has also been a genetic adaptation within the individual species. For example, Sanda (1978) showed that ecotypes of grass that grew close to the beach was more salt tolerant than the inland types. This indicates that coastal vegetation is damaged less than the inland vegetation under the same salt stress.

High salt exposure will change the individual species' competitiveness in a vegetation community, and species composition changes can be expected. L.B. Pedersen and Inger Slev (2007) refer to sources in order to describe examples of this. In Norway, for instance, found seaside plantain (Plantago maritima) along the E6 in Hedmark (P. A. Pedersen et al. 2002). However, it is not obvious that the occurrence of the species which normally grow in the coastal zone is due to increased salt levels. Many such species also act as weeds in arable land. More disturbance of the earth's surface in connection with road maintenance must also be assumed to be an important reason why a particular vegetation composition occurs near the road.

Wilcox (1986a, b) studied the effect of salt exposure from a salt storage place on marsh vegetation in Indiana, USA. Salt exposure had lasted for 10 years and resulted in almost all the local species disappearing and been replaced by immigrant vegetation dominated by narrow leaf cattail (Typha angustifolia). Salt levels were in this case very high, 468 mg/l Na and 1215 mg/l Cl in the peat-water. Similar levels were found in Norway in plant available groundwater near rv2 in Akershus (Røhr 1996). During the four years after exposure to salt ceased and the concentration decreased to half many local species had re-established, but several of the new species continued to expand. In laboratory tests Wilcox (1984) proved reduced growth of peat moss (Sphagnum palustre) at chloride levels as low as 300 mg/l in a water solution. Buzios, et al. 1977 also examined the effects of salt exposure from a storage facility for sodium chloride. 29 species invaded areas that had a soil concentration under 5000 ppm sodium chloride, whereas only 6 species invaded areas where concentrations were between 20000 and 25000 ppm. They argue that the species *Puccinella distans* (weeping alkaligrass) can serve as indicator plants for extremely saline areas. In a study of a wetland area in the United States, Richburg et al. (2001) concludes that road salting with sodium chloride resulted in the invasion of reeds (Phragmites) and reduction of species number. Concentrations in the soil solution were up to 390 mg/l Na and 275mg/l Cl which are equal to levels recorded in strong salt affected areas in Norway (Røhr 1996, Pedersen et al. 2002).

6.10 Interaction Effects

It is mentioned earlier how sodium chloride can affect plants, both primarily and secondary. They secondary effects are partly described as soil chemical changes and deterioration of soil structure (see Chapter 5).

Exposure to salt may also result in increased attacks by harmful organisms. Braun and Flückiger (1984) found increased susceptibility to an attack of apple aphids on hawthorn trees after they were sprayed with sodium chloride. Sodium chloride was used in 20 and 100g per liter concentrations in the water solution, which is more than in seawater (here in other words, it is the study of mechanisms that are important). They found increased amino acid content, including aspargine and glutamine and increased sugar content that they believe may have resulted in better conditions for the aphids. Trockner and Albert (1986b) also proved increased glutamine content in salt damaged trees, but overall reduced amino acid content. Braun and Flückiger (1984) address, several other studies with a correlation between pest attack and salt exposure. In Norway, several instances of salt damaged spruce populations are observed to be affected to a greater extent by bark beetle attacks than undamaged stock in the immediate vicinity (P.A. Pedersen and Fostad 1996).

Fungi attacks on saline-exposed trees have also been recorded. Northover (1987) found positive correlation between sodium and chloride and fungi attacks on roadside peach trees. In certain cases it may conceivable that salt damage provides entry opportunities for harmful fungi (Gibbs and Burdekin 1983).

Salt stress seems to affect winter hardened plants. Highly reduced salt tolerance has been found in wheat plants that have been exposed to salt stress (Tyler et al. 1981).

6.11 Discussion

Critical load for salt can be defined by correlating growth disorders or the extent of symptoms to concentration of chloride or sodium in leaves, soil, soil solution, nutrient solution and irrigation water. Critical loads may also be related to damage symptoms by applying a certain dose of sodium chloride to the soil. It is important to note that chloride is far more mobile in soil than sodium and that chloride levels vary greatly throughout the year and between years. Low levels of chloride in the upper soil layers does not necessarily mean that exposure has been small, rather that the leaching has been great. In Pedersen and Fostad (1996) and Pedersen et al. (2002), several examples of the dispersion curves for chloride in soil are shown and are far more irregular than the dispersion curves for sodium.

It is problematic to determine the critical loads that are applicable in a given field situation because the extent of damage depends on several factors. Symptom development and concentration change in plant tissue can take place differently over time and the relationship between them will therefore depend on the timing of sampling (Sanda 1976, Pedersen PA 1990, PA Pedersen and Fostad 1996). It will take some time from harmful concentrations to visible symptoms. This represents, however, an even greater uncertainty of measurements of salinity in the soil, especially in short-term experiments. Fostad and P.A. Pedersen (2000) demonstrated that damage amongst species such as spruce (Picea abies) and birch (Betula pendula) after sodium chloride application increased over a period of 12 months. It must also be taken into account that the genetic variation in salt tolerance within species is significant. Relating extent of damage to concentrations in soil solution is not problematic because these concentrations are dependent on the water saturation in the soil. A Norwegian field study showed a steadily decreasing concentration trend (Røhr 1996), whilst the extent of damage to Norway spruce at the site was continuously increasing (P.A. Pedersen and Fostad 1996). In a cultivated experiment with common spruce and pine, serious damage to the spruce planted in a growing medium (peat), occurred at respectively, about 500 mg Cl/l peat and about 600 mg Na/l peat. The pine were somewhat damaged at these levels, but only showed the same extent of damage as spruce at concentrations in the soil of approximately 1200 mg Cl/l peat and about 100 mg Na/l peat (Fostad and PA Pedersen 1997b). In forested areas in Norway where spruce is obviously damaged, the levels of sodium and chloride in the upper soil layers are approximately 100 to 400 mg/l soil (P. A. Pedersen and Fostad 1996). However, as mentioned earlier, the levels can be higher in the deeper soil layers in such locations.

Since salt uptake is dependent on the transpiration rate, and at the same time salt concentration in the leaves decreases with increasing growth, an accurate determination of the critical load seems to be difficult.

Horntvedt (1975) provides in its literature an overview of chloride content in leaves related to the extent of damage in several species. Generally concentrations higher than 0.5-1% dry plant material seem to yield significant damage depending on, among other things the species. In non-exposed trees the content is normally at 0.1% or lower.

P.A. Pedersen and Fostad (1996) found certain correlation between chloride content and extent of damage on spruce in general, but there is still a large overlap between the different damage classes (Table 3). These trees were exposed to both spray and salt in the soil and the analysis does not reveal how much of the salt that lay outside the needles. In the same study, they found that the extent of damage at approximately 1% chloride ranged from negligible to great damage, in deciduous species. Bäckman (1980) found that the needles from damaged spruce contained 0.5% chloride or more. Kayama et al. (2003) found clear correlation between the needles survival and their content of sodium and chloride. At levels of approximately 150μmol/g dry matter chance of survival was greatly reduced. Corresponding levels in the needles (0.5% of dry matter) caused great damage to common spruce in the Norwegian trials (Fostad and P.A. Pedersen 2000).

Table 3: Correlation between visible damage to the common spruce (Picea abies) along salted roads in Østlandet, Norway and the content of sodium and chloride in the needles (%) 0 = undamaged, 9 = destroyed (P.A. Pedersen and Fostad 1996)

Number of samples	Damage (0-9)	Na (%)	Cl (%)

24	None (0)	0.07 (0.02-0.10)	0.16 (0.06-0.36)
13	Very weak (1)	0.12 (0.05-0.22)	0.14 (0.09-0.27)
11	Light to moderate (2-3)	0.17 (0.05-0.37)	0.30 (0.10-0.54)
7	Strong (4-8)	0.19 (0.08-0.36)	0.49 (0.24-1.05)

As mentioned in chapter 6.9.1 and 6.9.2 sodium chloride damages plants, both by uptake through the roots and by direct spray/splash. There is not necessarily any correlation between the tolerance to direct salt spray and uptake through the roots. Thompson and Rutter (1986a, b) have demonstrated this through controlled spraying and irrigation with sodium chloride. Norway Maple (Acer platanoides) seems to be tolerant towards salt spray (Lumis et al. 1976), but can withstand very little salt in the soil (Fostad and P.A. Pedersen 2000). As mentioned, in several investigations it is established that the addition of air-transported salt decreases sharply with increasing distance from the road and that the salt spray damage to vegetation, thereby also decreases when the distance increases (Blomqvist 2001 with others). The extent of salt spray damage appears to be dependent on climatic conditions during exposure (Viskari and Karenlampi 1999). Critical loads may be set at distance intervals where different degrees of damage are assumed to occur at certain salting procedures and traffic patterns. Local variations seem to be very large (P.A. Pedersen 2007) and can make it difficult to predict how much damage may occur.

Registrations of salt damage along the Norwegian roads and in laboratory experiments show large differences in tolerance among pine (Pinus silvestris), spruce (Picea abies) and birch (Betula pendula) in terms of salt uptake through roots. However, when regarding spray above ground all three species are rather sensitive. Direct spray damage is common in pine and birch, but root uptake rarely results in any major damage. Horse Chestnut (Aesculus hippocastanum), however, seemed to withstand a lot of salt spray, probably because the buds are covered with a resin-like substance (Lumis et al. 1973). This corresponds to the Danish observations in winter 1995/96 where salt spray damage was extensive, but Horse Chestnut coped quite well (Olsen pers. comm.) and experiments with controlled salt application (Paludan - Müller et al. 2002). However, in practice Horse Chestnut is often damaged by a high salt content in soil (Sanda 1976, Fostad and Vike 1991). In urban environments the speed limits are low and risk for salt spray damage is minimal. Salt damage due to uptake from the soil is a serious problem. In such an environment it may take a long time before the symptoms of damage cease even if salting ceases. This is probably due to delayed leaching of salt due to permanent cover such as asphalt and stone. In Denmark, mats are used in winter months to protect the verges against salt spray, but studies by L. B. Pedersen et al. (2000) suggest that the effect of such measures is relatively small. Control of runoff from the road is probably a more effective measure.

Effects on larger areas and ecosystems will, as mentioned in the previous section, depend heavily on local conditions related to soil type, topography, rainfall and wind. Based on the results of the interdisciplinary project by Public Roads Administration in the nineties and later the follow-up (P.A. Pedersen and Fostad 1996, Røhr 1996, Fostad and P.A. Pedersen 2000, Pedersen et al. 2002) it can be suggested that salt in certain locatilities may accumulate and the extent of damage increases over time. This happens where the vertical transport of salt is ineffective and the precipitation height is low. At such locations critical loads for several species are exceeded with respect to uptake from the soil, under the actual salting procedures that have been followed. In such areas significant horizontal transport is observed, and there is an observed example that an open trench has had a considerable effect on the salt level in the upper soil layers (P.A. Pedersen and Fostad 1996). It is also shown in several cultivation experiments that the plants are more susceptible to salt damage if they are grown in sandy soils than if they are grown in more fine-grained soils or organic peat (Sanda 1977, Townsend 1984, Fostad and P.A. Pedersen 2000). In the field, however, the salt is washed out of the course-grained soils the easiest and serious damage as a result of root uptake is less likely. The consequences of high salt application are very different along the coast and inland. In coastal areas precipitation is generally high and contributes to the salt being washed effectively. Inland, rainfall is low and there are long periods of rainfall deficit, i.e. evaporation from the ground and transpiration from vegetation is greater than rainfall. Inland areas will therefore be sensitive to extra salt application. In Norway it is proved that the salt level in the soil along salted roads in Østlandet has increased during the season while it decreased in Rogaland (P.A. Pedersen and Fostad 1996).

The genetic variation that is found in respect to salt tolerance between and within species suggests that coastal vegetation is better adapted to high salt levels than inland vegetation. Salt damage to trees along the coast is due largely salt spray, and such damage is not necessarily accompanied by high levels in soil. Since sea spray often causes damage to native vegetation along the coast and many of the species that naturally grow along the coast can not be considered particularly salt tolerant, it indicates that salt exposure is not great enough to cause strong selection. Salt exposure near the road along the many road stretches is probably much greater than along the coast, and it will hardly be possible to avoid damage to the vegetation along roads by using coastal ecotypes unless one plants very special species that grow down to the shore and that are adapted to high salt content in the soil.

Changes in species composition are a natural consequence of road salting. The focus of some studies is the changes in vegetation communities as a result of salt exposure of soil and ground water (for example, Wilcox 1986a, b). The major damages found on spruce caused by salt in soil (Fostad and P. A. Pedersen 1996, P. A. Pedersen et al. 2002) suggest that spruce would be removed along some stretches of road. Salt spray from the road will also be expected to affect species composition locally. Since salt spray damage is most extensive near the ground (but above the snow cover) the start-up phase will be critical for sensitive tree species such as birch and pine, up to a certain distance from the road.

Alternative de-icing chemicals will also have effects on plants. The chemicals are however very different and will have different influence mechanisms. Organic agents affect the oxygen conditions in the soil when they decompose (Chapter 5.2). Alternative inorganic salts will necessarily affect nutrient uptake and solubility of plant nutrients in the soil during ion competition and ion exchange on the soil colloids. Documentation of the effects of alternative chemicals is currently limited.

6.12 Effects of other de-icing chemicals on plants

Robidoux and Delisle (2001) examined the effect of sodium chloride, sodium formate and calcium magnesium acetate on earthworms (Eisenia fetida) and four plant species: cress (Lepidium sativum), barley (Hordeum vulgare), and grass (Festuca rubra and Poa pratensis). They found that calcium magnesium acetate was somewhat less toxic than sodium formate, which was about as toxic as sodium chloride. Effects on the plants (with Poa pratensis as the most sensitive) were greater than those on the earthworm. The results from experiments show that the average concentrations of sodium chloride found in the snow (about 10000mg sodium chloride/l) had effects on both grass species, as well as cress, while the barley was not affected by these concentrations. Concentrations of sodium chloride in old snow (up to 30000mg sodium chloride/l) however did result in effects on seedlings of barley.

It will then be necessary to use larger amounts of NaFo and CaMg-acetate to obtain the same salting effect; Robidoux and Delisle (2001) assume that the effects of these de-icing chemicals on plants and earthworms will be about the same as for sodium chloride.

In a lysimeter experiment conducted with potassium formate all plants died probably as a result of high pH in potassium formate solution (pH 10.6-11.4) (Hellstén and lately in 2003).

Fischel (2001) provides an overview of studies where the effect calcium magnesium acetate is tested on vegetation. Results show that the effects of calcium magnesium acetate on most plants (trees, shrubs) were few or negligible at doses that are commonly in use. In studies where the effects of calcium magnesium acetate are compared with the effects of sodium chloride, it seems that the effects of calcium magnesium acetate are considerably less.

6.13 Conclusions and Recommendations

Sodium chloride causes obvious damage to shrubs and trees along salted roads and streets. Damage includes reduced ornamental value, growth and survival. Symptoms and extent of damage varies with exposure, environmental conditions at the site and the salt tolerance of the individual plant in question. There is often no connection between the individual species' tolerance to salt spray and tolerance to salt exposure by root uptake. The species that grow along the roads in Norway, both spontaneous and planted vegetation, are generally relatively sensitive to high salt levels, but still display considerable variation.

It is therefore recommended that wherever possible plants are used which can tolerate increased salt levels and that when selecting the species it is taken into account the type exposure that is dominant on the site. Further documentation on the effects and the choice of salt tolerant species used along salted roads is needed.

The extent of salt damage generally decreases with increasing distance from the road (especially salt spray-injury), but significant damage can be expected at large distances (more than 50 m) in some situations. One should avoid the planting of species that are sensitive to salt spray closer than about 10-15 m from a roads edge where the speed limit reaches 80-100 km/h. It should be taken into account local conditions in the planning of green areas, and when upgrading green areas in the plantation near the road to the experience and thorough knowledge of place is necessary. The reasons for the large variations in damage due to salt spray should be documented better.

In cities the speed limits are often low and the damage is caused mainly by salt uptake via the roots. Generally, it is recommended to plant in higher verges so that salt runoff directly to the root zone is avoided. Controlled directing of surface water from road salting in the season is an important management measure. In planning surface water management from urban centers the transport of salt to the trees root zone should be in general included as a problem.

The extent of damage to forest vegetation along salted roads depends heavily on local soil conditions, rainfall conditions and slope conditions in the terrain. A common feature for sites with a large extent of damage is that there is often a dense layer in the soil that prevents leaching of salt and the amount of rainfall is relatively small. Soil studies show that the salt found under such conditions is dispersed to larger distances downstream of the road and results in high levels in the trees root zone up to 100m from the road. Under these conditions, the spruce will be ousted and there will be major changes in species composition.

To reduce the salt impact of these sensitive areas the surface water should be directed away from the area in a ditch system with a good drop. Special design of road ditches with the sealing of trench sides can be applicable to control direction of water in the best possible way. Additional trenching outside the road ditch may be a possible management measure used to intercept the horizontal salt transport out of the side terrain. The problems in these areas have been shown to increase over time, and continuous surveillance of selected areas should be carried out in order to document developments over time.

Harmful effects of the alternative de-icing chemicals on the plants have also been proven, and more thorough documentation of these effects is required.

7. Effects of de-icing chemicals on domesticated plants

7.1 Roadside damage caused by sodium chloride

Salt damage along the roads is frequently reported for trees and shrubs, and a distinction is not necessarily made in the literature between crops and plants in the forest environment and cultural landscape. Possible effects of road salting on agricultural production are not often examined. Persson and Røyseland (1981) examined the effects of road salting on vegetable production along roads in Vestfold, Norway, but could not document any reduced growth near the road. The U.S. report Berkheimer et al. (2006) stated that salt spray from the road is a likely cause of extensive shoot death of large fruited blueberries (*Vaccinium corymbosum*). Eaton et al. (1999) also showed damage to flower buds and yield reduction of blueberries along a salted road (*Vaccinium angustifolium*).

In arid and semi-arid areas (such as Australia and some Mediterranean countries), the high salt content in soil and irrigation water can be a significant problem in agriculture. The reason for this is that salt and nutrients supplied by rainfall, fertilizer and irrigation are not washed out from the earth, but instead accumulates in the upper soil layer (Brady 1984). The flow of water in these areas will often go towards the surface due to evapotranspiration. In areas along the salted road, it is proven that the content of sodium chloride in the soil, groundwater and surface water may increase as a result of no salt (see respectively chapters 5.1, 10 and 11). Indirect damage to crops due to poor soil structure and water management, as well as burn damages on crops as a result of irrigation with water that has a high content of sodium chloride because of road salting can not be ruled out. Respective effects on soil and plants as a result of the use of sodium chloride and other de-icing chemicals are described in Chapter 5 and 6.

7.2 Relative salt tolerance of domesticated plants

High salt content in soil can reduce water uptake, reduce the plants roots, lead to burn damage, inhibit flowering, reduce seed germination, and reduce crop (fruits and vegetables). Shown below is an extract of an overview of salt tolerance for different cultural crops which was designed by Colorado State University (Table 4). Such a list only gives an indication of relative sensitivity between the species. Local climatic conditions and soil conditions may change part of this picture. The conductivity values given (Table 4) are from the conductivity in the soil. In other words the overview gives no information on how plants can withstand salt water used for irrigation (or eventual burn damage).

Table 4: Overview of the salt tolerance of some cultivated plants. For a complete list see http://www.ext.colostate.edu/mg/files/gardennotes/224-SalineSoils.html

Non-tolerant (0-2mS/cm)	Some tolerance (2-4 mS / cm)	Moderately tolerant (4-8 mS / cm)	Tolerant (6-16 mS / cm)
Carrot	Apple	Broccoli	Asparagus
Onion	Cabbage	Tomato	Beet

Peas	Celery	Spinach	Olive
Radish	Cucumber	Squash	Common Jupiter
Strawberry	Salad	Ryegrass	
Pine	Potato	Chrysanthemum	
Sugar cane			

Relative salt tolerance (based on salt content in soil) between grain species indicates that barley is more tolerant than oats and wheat (for example, Carter (1981). We have not found similar overviews of other relevant chemicals used during winter maintenance of roads.

7.3 Discussion and conclusions

Salting (sodium chloride) of roads takes place at a time of year where food and animal feeding crops are not grown so that the likelihood of splash damage from irrigation water that contains road salt is small in Norway.

Effects on the domesticated plants grown along the roads as a result of changes in soil structure and nutrient balance in soil can not be excluded (see Chapter 5 for description of such effects). Arable land is regularly harvested or ploughed so that the salt that is added is mixed into a larger soil volume than in the soil where such processing does not occur. When soil processing will help to dilute the added sodium chloride, it is likely that this will help to reduce short-term effects of salt. At the same time both fertilizer and lime are applied to arable land, so that some of the effects observed in soil where sodium chloride is applied will become smaller or disappear entirely.

There are currently no studies on how de-icing chemicals used during winter road maintenance affect the arable land in Norway, it is therefore difficult to estimate what effect this has on such soil quality and crop yield over time.

8. Effects of sodium chloride on soil fauna

8.1 Introduction

Road dust from 7 Norwegian cities (28 samples) were analyzed chemically and physically and a number of ecotoxicological tests involving worms, springtails, bacteria and plants were carried out on the road dust (Amundsen et al. 1999). One of the main conclusions was that it was not possible to associate observed effects to any/some contaminants, but that the observed effects were caused by a combination of influences from many contaminants. However, there was a tendency that the test mixtures with chloride content larger than 60mg/kg (+ other contaminants) resulted in significant inhibition of lettuce germination, as well as survival and reproduction of earthworms and springtails. Salt was the only chemical factor in the road salt that could be correlated roughly to the biological effect.

Soil organisms have a variety of mechanisms that enable them to survive periods of unfavorable living conditions in the soil: they curl up like a ball (reducing the surface), become inactive, they stop eating and stop reproducing. This is one of the reasons why the mortality of organisms is a parameter which is less useful for measuring the effects of chemicals. Measurements of sub-lethal and chronic effects occur often at lower concentrations than acute effects (mortality). This is evident for both organisms in soil and water.

8.2 Effects of sodium chloride on soil invertebrates

Studies of the effects on soil living organisms are not performed to the same extent as in water. The most comprehensive study of the effects of salt on soil organisms that we have found reported is by Addison (2002). Here the effects of sodium chloride on soil living organisms studied (4 species of earthworms, 6 species of springtail). Among these organisms were springtails (*F. candida*) the most sensitive and a 50% reduction in reproduction was found at concentrations from 480 to 940 mg of sodium chloride/kg (depending on soil type and water content in soil). For earthworms (*E. fetida*) for example a 50% reduction in cocoon production was measured at concentrations of approximately 1800 mg sodium/kg soil.

The tests show that the effects of deicing agents are dependent on soil type. For example, the effects are reduced at an increased content of organic matter in soil.

Mortality in earthworms and springtails (measured as LC^{20}) was detected at significantly higher concentrations (3000-15000 mg sodium / kg soil) (Addison 2002). This indicates that mortality is an insensitive endpoint for the negative effects of road salt.

Another observation made was that pure sodium chloride was more toxic than road salt (probably sea-salt) for some species (Addison 2002). This effect is also seen to aquatic organisms and explained by that a wide range of other cations and anions are found in road salt (Ca^{2+}, MG^{2+}, SO_4^{2-}), which will reduce the effect of Na and Cl. If other ions are available in the liquid phase forming these together with Na and Cl is probably a more "physiologically correct" composition of the water phase. The presence of Ca, Mg etc. can also lead to reduced toxic effects as a result of complexing reactions with chloride.

De Barros Amorim et al. (2005) found in experiments studying the effect of $CuCl_2$ on

Enchytraeus (small white roundworm that live in soil) and springtails (*Folsomia candida*), that the chloride ion had an effect on the reproduction of these animals in some soil types.

Bongers et al. (2004) found that Pb-nitrate was more toxic than Pb-chloride for springtail *Folsomia candida* in the soil. After the soil which was added to the Pb-salts was washed with water and the anions washed out, there was no difference in toxicity between these two salts.

Schrader et al. (1998) found that concentrations of 2400 mg sodium/kg soil, or 2800mg $CaCl_2$/kg soil had negative effects on egg development of *Poecilus cupreus* (beetle) and *Folsomia candida* (springtail).

The results from these experiments can not be used to estimate at which concentrations the effects of chloride occur. The clear effect chloride has on several soil living organisms suggests that the use of chloride salts in toxicity testing (for example, the use of $PbCl_2$, $CuCl_2$, etc) control tests must be conducted to separate the effect of chloride from the effects of the toxic cations (this can be done for example, by leaching of chloride, or by using KCl as a reference).

8.3 Critical loads of sodium chloride in soil fauna

Environment Canada (2001) reference studies that show that the sensitive bacteria on roadsides become moderately inhibited at sodium chloride concentrations of 150mg/kg (equivalent to respectively 60 and 90 mg Na and Cl/kg). Nitrification in soil is affected by concentrations of 250 mg/kg. In Canada, it is shown that concentrations of sodium chloride in the soil closer than 30m from the road has a are higher than 60mg/kg, while concentrations of chloride higher than 200mg/kg have been recorded as far away as 200 m from the road. This shows that micro-organisms in large areas of soil around heavily trafficked roads can be affected by salt use.

It is unclear what the critical load of chloride for soil fauna is. Chronic effects of sodium chloride are detected at concentrations as low as 480mg of sodium chloride (or 280mg Cl/kg) (springtails), while earthworms have been shown to be somewhat less sensitive.

8.4 Discussion and conclusions

As for the organisms in the water, chronic and sub-lethal effects of salt occur at concentrations that are significantly lower than the concentrations of acute effects. When conducting studies on effects, it is therefore important that the chronic and sub-lethal effects are measured and not just acute.

By using organic de-icing chemicals, decomposition in the roadside soil is important for the environmental effects that may occur. Micro-organisms are responsible for most of this decomposition and these will adapt to growth conditions and the substrate (deicing chemical) which is applied on site. In a well-functioning ecosystem microflora, fauna and flora depend on each other and it is unclear whether or to which extent the cleaning ability of the soil will be reduced if some parts of the ecosystem disappear.

If one bases the decomposition of organic de-icing chemicals on infiltration in the soil (and not collection and purification in ponds), it is important to determine which soil types could act as a cleaning medium and which are inappropriate.

9. Other ecological effects of sodium chloride

9.1 Effects on mammals and birds

Sodium chloride can have both a direct (toxic) and more indirect effects on wild mammals and birds.

An important effect road salting may have on terrestrial animals is that salt attracts animals who need it in their diet (Hedin 2006). The animals are thus more prone to accidents. Research from Ontario, Canada, shows that collisions between cars and animals are greatest in the period where the need for salt is greatest, not in the period which motor vehicle traffic is highest. Research cited in Environment Canada (2001) also shows that there are many more collisions in areas where saline surface water (due to road salting) exists in relation to other areas (regardless of traffic density). This applies primarily to larger animals (moose, deer, etc), but other research indicates this may be the case also for smaller animals.

Birds are attracted by the environment around a road because food is found readily available (dead animals and insects) and sand, but also because the salt that birds need are found in certain periods. Mineau and Brownlee (2005) discuss the results from multiple studies or reports on bird deaths in connection with road salting. A study was referred to where the acute toxicity of sodium chloride to the House Sparrow was investigated. Endpoints such as mortality or changes in plumage, electrolyte levels in brain and plasma were measured after feeding with sodium chloride. The lethal oral dose was found to be 3-3.5g/kg body weight and a possible no-effect level was estimated at 2g/kg body weight. If the birds were kept away from water for 6 hours after exposure, the mortality increased. Mineau and Brownlee (2005) believe that the intake of salt on some roads and for some organisms is in general an underestimated problem.

Environment Canada (2001) also refers to several studies showing that there is a connection between the salting of roads and mortality of birds. It isn't stated to what extent this mortality is significant for the population, although it refers to species that are protected.

Fischel (2001) gives an overview of the toxic oral dose (lethal dose LD_{50}) for rats of different de-icing chemicals (Table 5). Magnesium chloride is the least toxic substance, while potassium acetate is the most toxic. According to the references given in Fischel (2001) values lower than 5000 mg/kg are considered to be non-toxic.

Table 5: Acute oral toxicity to rats for selected de-icing chemicals (cited in Fischel 2001).

Deicing agents	LD_{50} (mg/kg)
Magnesium Chloride	8100
Calcium Magnesium Acetate (CMA)	5000
Calcium Chloride	4000
Sodium Chloride	3750
Potassium acetate	3250

9.2 Changes in species composition

The chemical and microbiological changes that occur in soil and water along the road as a result of salting may also lead to changes in species composition in plant communities along the road (Båtvik et al. 2001). This is shown in Canada amongst others where it is believed that road salting, trenching, combined with fertilization in agriculture has led to reeds (Phragmites) spreading rapidly in some areas (Jodoin et al. 2008).

In some areas salting can lead to increased erosion, both because the plants die and because of poorer aggregate stability due to leaching of Ca and Mg. In such areas an increased quantity of suspended substances can cause acute or chronic effects in the water and it can cause future changes in species composition.

9.3 Discussion and conclusion

To which extent the use of de-icing chemicals on roads means an additional problem for terrestrial animals and birds is not clear, although some birds can apparently take in lethal doses of sodium chloride. We believe it is likely that any "additional burden" caused by the use of de-icing chemicals is very small in relation to the burden the road already causes these animal groups. Further investigations of the problem should not be given priority in the work on the development of salt strategies in Norway.

10. De-icing Chemicals in the groundwater

In this chapter some general information is compiled together based on textbook material and references are not necessarily included here. More concrete examples and results of research are described with references to the literature.

10.1 Environmental Implications

Which biogeochemical interactions are affected by the addition of de-icing chemicals in soil and groundwater generally (unsaturated and saturated zone) are described in chapter 5. This section focuses the consequences in the saturated zone and particularly in relation to the use of groundwater as a water source.

Different de-icing chemicals will have different environmental impacts; the following criteria can be set up:

- Suitability for drinking (toxicity)
- Oxygen conditions
- Corrosive properties

10.1.1 Suitability for drinking

Drinking water regulations have provided a number of limits that may be of relevance for the assessment of water quality in connection with the use of different de-icing chemicals (Table 6). Threshold value for the Norwegian drinking water is 200 mg/l for both Na + and Cl-, this value gives a salty taste to the water. For persons with low-sodium diet this limit is too high. The recommended level for sodium is set to 20-25mg/l (the recommended limit in Norway). Limits may vary from country to country (Brod, 1998). This is because it is relevant to look at elements other than the actual de-icing chemicals and chemicals will be broken down into other substances which for example can be measured as TOC, or chemical ion exchange reactions may occur which leach out other elements from the soil, as described earlier. Iron and manganese are elements that typically occur at higher concentrations where there is a breakdown of organic matter. Although drinking water regulations set important limits, they should not to be considered as critical limits in relation to groundwater environment as a whole. In relation to the assessment of outlet areas for groundwater (springs), Williams et al. (1999) proposed a biological index system to evaluate organisms critical limits for chloride.

Table 6: Limiting Drinking Water Regulations.

Elements	**Unit**	**Relevance in relation to the use of the following de-icing chemicals**
Chloride (Cl)	200 mg Cl/l	NaCl
Sodium (Na)	200 mg Na/l	NaCl
Glycols	10 µg C/l	Use of glycols (ethylene, proylen-) these are already used in cars during the winter
Chemical oxygen demand (COD-Mn)	5 mg O/l	Organic de-icing chemicals
Conductivity	250 mS/m	Organic de-icing chemicals and NaCl
Manganese	0.05 mg	Organic de-icing chemicals

	Mn/l	
Nitrite (NO_2)	0.05 mg N/l	Urea
Nitrate	10 mg N/l	Urea
Total Organic Carbon	5 mg C/l	Organic de-icing chemicals

10.1.2 Oxygen demand

Oxidation processes such as decomposition of organic de-icing chemicals (propylene glycol, potassium acetate, potassium formate, carbohydrates) and turnover of urea to nitrite and nitrate, leads to a lower oxygen concentration in the groundwater. This results in a reduction of iron and manganese, which is water-soluble in this form. Water with high iron and manganese values can not be used for water supply unless it is oxidized in advance. High values of Fe (II +) and Mn (II +) in groundwater is undesirable because: 1) oxic conditions will lead to precipitation of iron (Ferric) and manganese (manganic) and will give the water a reddish or brown/black color, 2) such precipitates can give water a metallic taste, 3) the same conditions which release iron and manganese in ground water can also release hydrogen sulfide, which may give the water a taste of sulfur, 4) precipitation of iron and Manganese oxides can clog pipelines. More about these processes and measures to treat groundwater with reducing conditions are described in Søvik (2003) amongst others. NaCl will not have consequences in relation to the level of oxygen in the groundwater, but can affect the pH value. At Oslo Airport, Gardermoen (OSL), the chemical oxygen demand (COD, see Table 1) was an important criterion for selection of de-icing chemicals on the runways. Because of an observed increase in iron and manganese values in the unsaturated and saturated zone, OSL went from using potassium acetate to using potassium formate after only a few years of operation (Øvstedal pers. comm).

10.1.3 Corrosion

Different de-icing chemicals can increase the corrosion ability of the water. Sodium chloride seems to be the most corrosive followed by urea and organic de-icing chemicals. Anti-corrosion agents are often added to commercial deicing agents (for example see Fischel 2001).

Corrosive properties of de-icing chemicals are not mentioned here.

10.2 Transport properties of de-icing chemicals

Generally, the transport properties of different elements depend on; the charge of the item, affinity for different soil types, reaction with other substances in the soil, for example complexing, solubility.

The degree of binding in soil is described often with a coefficient (Kd value) which is dependent on the soil type and salt concentration in the water. This is an important parameter to know if one will calculate how fast an element will move compared with pure water. The various salts as described will occur in the form of ions in the water. The positive ions, Na^+, K +, Mg^{2+} and Ca^{2+}, will be delayed more than the negative ions because the soil has a greater cation exchange capacity than an anion exchange capacity. The components Cl-, Acetate and Formate will therefore be transported at a rate nearly equal to that of water (for examples of the transport rates of potassium, acetate and PG see French et al. 2001). It is possible to find Kd values in the literature for similar soil types, but there are also simple methods developed to measure this in a laboratory trial (batch and column experiments). More general information about the processes and methods can be found for example in Appelo and Postma (1996).

10.3 Use of models in conjunction with the dispersal studies

In order to model the dispersal of de-icing chemicals from the road it is necessary to survey all mechanisms that influence the spread (wind, spray, snow ploughing, runoff and further transport of unsaturated and saturated zone (Figure 2).

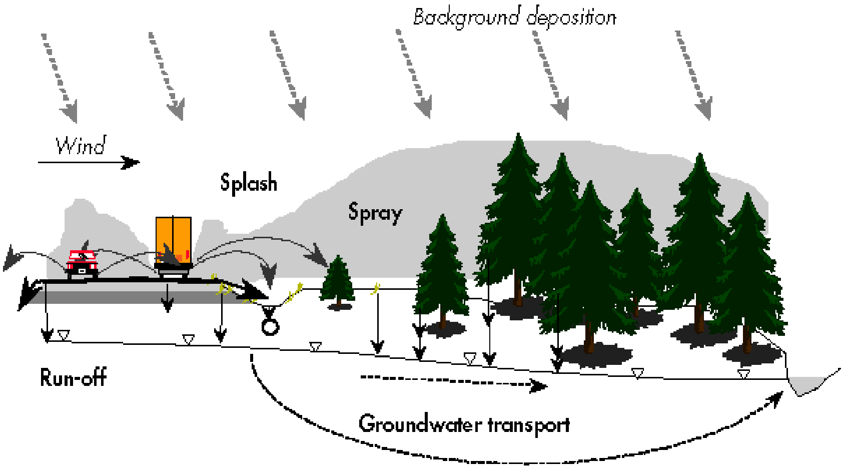

Figure 2: Spreading mechanisms for deicing agents from the road to the environment (Blomqvist 2001).

The purpose of models may be to:

- Optimize salting in relation to the friction requirements and bare-road strategies.
- Optimize salting in relation to not exceed the "tolerance limits" to the surrounding environment i.e. plants, animals, requirements for drinking water etc.

In both cases it will be necessary to calculate the mass balance, which means accounting for the amount of water and de-icing chemicals. The mass balance of salt and water can be calculated by how much water is available to mix with the salt until it reaches a sensitive recipient. In the following sections different types of models are described, and examples of some of these models are presented.

10.3.1 Dispersal from the road

Three transport mechanisms are important for salt dispersal from the road:

- dispersal through the air (splash and spray)
- dispersal by snow ploughing
- dispersal by runoff

These dispersal mechanisms have complicated impacts on the environmental factors that are affected and how further spread in the soil and groundwater takes place.

The following equations can be set up to describe the various transport components (Lundmark, 2008):

$$q_{Cl_a} = k_a \cdot S_{Cl} \qquad S_s < S_{pm}$$
$$q_{Cl_a} = k_a \cdot S_{Cl} + k_p \cdot S_{Cl} \qquad S_s > S_{pm} \qquad \textbf{(Equation 1)}$$
$$q_{Cl_r} = k_r \cdot q_w \cdot C_{Cl}$$

Where S_{CL} is the salt storage (mg m^{-2}), q_{Cla} and q_{Clr} are respectively, the dispersal rate through the air and with runoff (mg m^{-2} d^{-1}), qw is the surface runoff (mm), C_{Cl} is chloride concentration. S_s is the amount of snow on the road (water equivalent, mm.), if the accumulation of snow exceeds Q then the road is ploughed. k_a, k_p, k_r (d^{-1}), are the spread coefficeints the air, ploughing and drainage.

There are different model concepts used to calculate the transport and dispersion of de-icing chemicals from a road to environment. Annika Lundmark studied both dispersal mechanisms in the air, in the runoff and on the surface in her doctoral thesis (Lundmark, 2008).

To calculate the dispersal zone along the road, Blomqvist (1999) proposed that the following empirically based model is used:

$$D(x) = a_{splash} \cdot e^{-0,5x} + a_{spray} \cdot e^{-0,05x} + a_b \qquad \textbf{(Equation 2)}$$

where D(x) is the dispersal function mg/m2/day as a function of distance from the road x, a_{splash} and a_{spray} are the maximum dispersal distances due to splash (larger volumes of slush/snow), and spray (dispersal of small droplets). In experiments conducted by Mark Lund, the following values were calculated: a_{splash} = 8000 mg/m2/day, a_{spray} = 120 mg/m2/day and the background deposition ab = 3 mg/m2/day. Moreover, it was estimated that the airborne dispersal described above corresponds to 45% of added chloride on the road. If this was stated in cumulative percentages the distribution would be 30% within the first meters of road, up to 90% up to 10m and the remaining 10% to 100m from the roadside. These constants are adjusted measurements taken along the E4 ca. 10 km north of Stockholm, where the salt load is about 8 tons of NaCl/km road. In a further developed version of model (2) (Gustafsson & Blomqvist, 2004), the wind speed, direction and number of passing cars are also included, but the equation is not fully documented and therefore not presented here, also see the publications Blomqvist and Gustafsson, (2006) and Blomqvist (2002), which is also a framework for the risk assessment to roadside vegetation. Lundmark (2003) presents in a literature study several examples of dispersal studies and similar distribution models that are built on the same principle as models described above. These are therefore not described in more detail here.

10.3.2 Dispersal through the soil and groundwater

Once the pollution has been added to the soils surface further transport begins either as surface runoff to the brook, and river water, or de-icing chemicals infiltrate the ground and will first have to flow through the unsaturated zone, where both air and water are found in pores. This gives good decomposition conditions because oxygen is more readily available than may be the case in groundwater zone, where all pores are filled with water. Flow direction in the unsaturated zone is mainly vertical; one can therefore argue that a one-dimensional model is sufficient. The addition of a pulse of pollution will result in a dilution with other water in the flow direction and across the flow direction; there is therefore also dispersal in the horizontal direction depending on how heterogeneous the soil is. In general, we say that the more variable the soil is the greater the dispersal which can be expected. In modeling and field experiments in the unsaturated zone at Gardermoen, which is heterogeneous coarse sand, the dispersal in the horizontal direction was about 40 cm after 1.5 m of vertical transport (French et al., 2001). In fine soils, the horizontal dispersal could be larger. If there is a large stretch of road where the strain along the road is high, the dispersal in the horizontal direction would have less impact on dilution in the unsaturated zone. Dilution also occurs before the strain impact of de-icing chemicals. A rule of thumb is that dispersal in the main flow direction is about 10 times higher than the dispersal across the flow direction (e.g. Domenico and Schwartz, 1998). Field trials and simulations in an unstructured but highly heterogeneous soil at Gardermoen showed ca. 5-20 times greater dispersal along the flow direction than across the flow direction depending on the infiltration rate and depth of groundwater (French, 1999). Numerical simulation of the residence times in the unsaturated zone shows that the most sensitive variable is infiltration rates (Kitterød, 2008). Heterogeneity leads to focusing effects (Kitterød 1997, French and Binley, 2004). This contributes to the transport speeds increasing while the potential for degradation is less (Kitterød, 2008, French et al., 2001). When de-icing chemicals eventually reach the groundwater, the main flow direction is horizontal, and it is also in this direction that the largest dilution will occur. If there are very high concentrations of salt (sodium chloride), one will only be able to get a density driven flow, and the salt will only sink down to the aquifer. If there are thresholds in the soil, areas with an accumulation of saline groundwater will occur, Nystén (1998) could not document such a sink in his studies, whilst a theoretical study by Niemi (1998) provides a general overview of which combinations of gradients and soil types where one can expect sink effects (Table 7).

Table 7: Situations in which salt water sinking can be expected based on various hydraulic gradients and conduction capabilities and an effective porosity of 38-47% (Niemi, 1998)

Sr = gravel — Sr —
Khk = coarse sand — KHk —
Hk = sand — Hk —
Hhk = fine sand — HHk —
Si = silt — Si —
HkMr = sandy moraine HkMr

$\frac{dh}{dx}$ \ K [m/s]	10^{-1}	10^{-2}	10^{-3}	10^{-4}	10^{-5}	10^{-6}
- 0,0001	-	++	++	++	+	+
- 0,001	-	-	-	+	+	+
- 0,01	-	-	-	-	-	-
- 0,1	-	-	-	-	-	-

- concentration maximum remains in the uppermost layer (no sinking)
+ long term salt input may cause density effects
++ density effects and sinking observed already after one year salt input

10.4 Mass Balance Models

A simple mass balance calculation is shown by Åhnberg and Knecht (1996), for a stretch of the E20 in Swedenby Brännebrona where there are a number of eskers. They used the catchment area ($3 km^2$) and drainage map for the area (250 mm/year) together with the annual salt consumption for a stretch of road within the catchment area (8 kg per m road, 1.5 km road), and

calculated an annual average concentration of 16 mg/l, in reality, on individual days measurements along the relevant area were of several hundred mg/l (Bäckman, 1980). It is not surprising that you get a discrepancy here, because the mass-balance calculations equalize the dilution of an entire year something which is far from reality. Although estimated average concentrations are far below those measured at the individual cases, the method is an easy way to record regional differences in potential impact. Thunqvist (2003b) proposes a further development of the methodology connected to a GIS system based on stationary conditions (i.e. no change over time based on a prolonged impact with annual average values).
Chloride concentrations in groundwater input, [Cl-]RE, can be estimated using this equation:

$$\left[Cl^-\right]_{RE} = \frac{M_{Cl}\sum\left(L_{WC}S_{WC}\right)}{M_{NaCl}\left(P-E\right)A} \qquad \textbf{(Equation 3)}$$

where A is the part-catchment area the road crosses, M_{CL} is the molecular weight of Cl, M_{NaCl} is the molecular weight of sodium chloride, L_{WC} is the length of the road that runs through part-catchment, S_{WC} is the salt supply for road on the stretch of road in question, *P* is precipitation, and *E* is evapotranspiration.

Moreover, the total chloride concentration [Cl-] DIS at the outlet for a larger catchment is calculated in addition to include the water that is supplied upstream, the equation is then:

$$\left[Cl^-\right]_{DIS} = \frac{\frac{M_{Cl}}{M_{NaCl}}\sum\left(\frac{m_{salt}}{\left(P-E\right)A}\right)_i A_i}{\sum A_i} \qquad \textbf{(Equation 4)}$$

where m_{salt} is the amount of salt in kg, and *i* refers to the partial-basin.

10.5 Statistical evaluation of vulnerable areas

Several Swedish studies (Lindberg et al., 1996; Gontier and Olofsson, 2003) have used a "Risk Variable Method "in a GIS framework. Gontier and Olofsson (2003) used the water sample data from about 100 wells within 150 m from the road in Småland in Sweden in order to highlight the risk zones along a road in Småland. That conducted a statistical analysis of the relationship between chloride concentration and proximity to the road, topographic factors, the height of the well, geology at the well, dominant geology of the area, well depth, etc. Chloride concentration in the wells is shown in relation to distance from the road in Figure 3. There is a great dispersal of data and it is not easy to see any clear relationship with distance from the road. By doing an ANOVA when analyzing the data a grouping of wells is found that are related to the factors mentioned above. The various factors (distance to roads, topography, etc.) was added to a GIS system and classified according to risk classes, the digital maps had a resolution of 50x 50 m. After these themed maps are classified according to risk, these are summed and you will see a risk map based on an overall assessment (Figure 4). Four risk classes are defined: very low risk, low risk, high risk and very high risk.

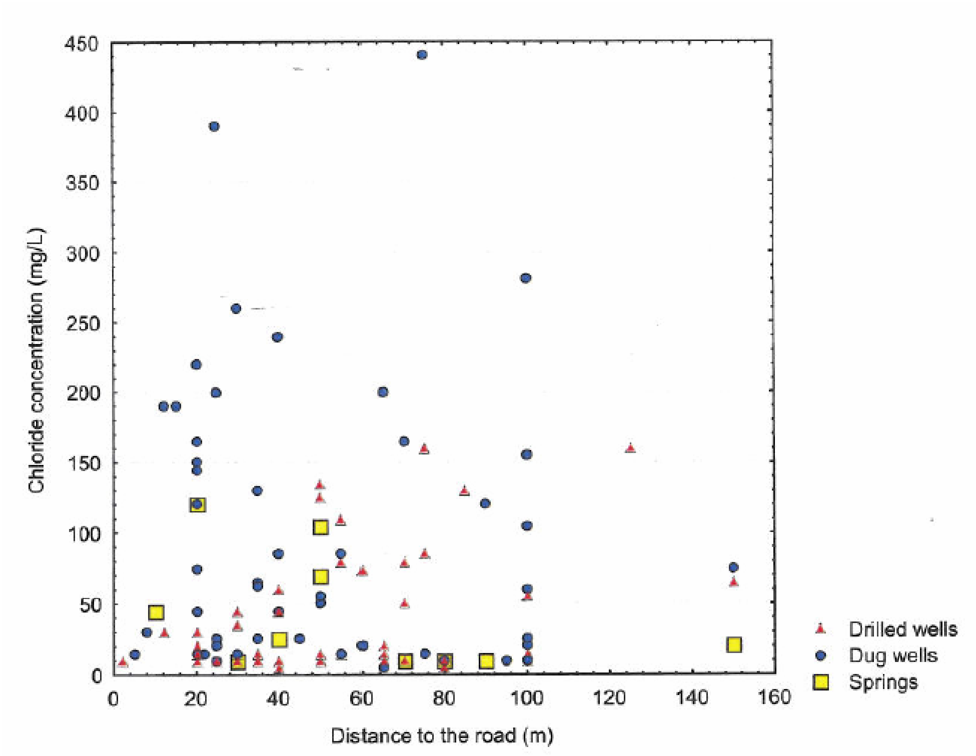

Figure 3*:* **Chloride concentrations in wells with different distance from the road (Gontier and Olofsson, 2003).**

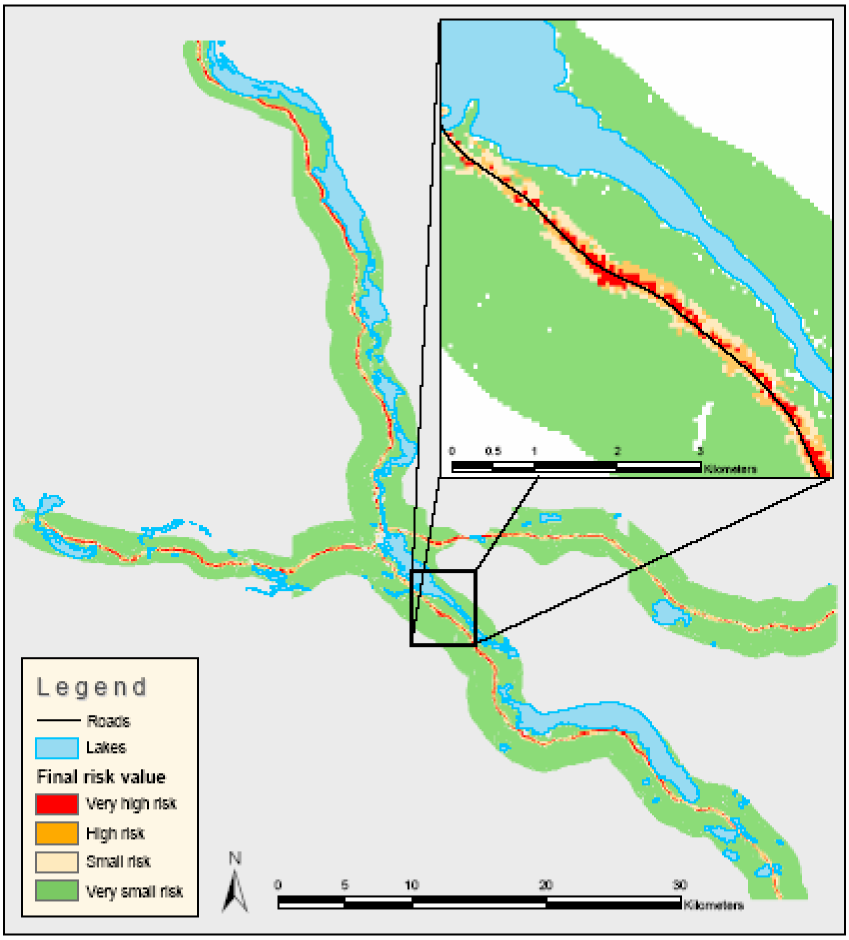

Figure 4*:* **Vulnerability map based on an overall assessment consisting of a conceptual review of factors and class division from digital databases (GIS) and an exponential distribution of salt from the road.**

In a similar risk tool developed in Finland (Kivimäki, 1994) the basis of environmental factors were used which had been proposed by Jones et al. (1992): area and volume of the aquifer, depth to groundwater, direction and speed of groundwater flow, soil type/geology, distance from the road to the aquifer, the road's drainage system, salt rates and volumes, volumes of water pumped out of the aquifer and extent of the salt layer in the projection area. The number of groundwater reservoirs of significance were counted within the various sub-areas in Finland and these were classified according to the sum of the risk factors mentioned above. The monitoring program is reported in Finnish, but there is also an English summary: http://www.ymparisto.fi/download.asp?contentid=70810&lan=fi Otherwise publications from the Finnish environmental research institute (The Finnish Environment Institute, SYKE) on this topic can be found here: http://www.ymparisto.fi/default.asp?node=11902&lan=en

A good framework for risk assessment in relation to pollution is described by Rosén (1998), see Figure 5. This article illustrates how the use of simple models (such as analytical solutions described later) can create sensitivity reviews/uncertainty analyses in relation to the choice of parameters and how this can be used in the further risk assessment.

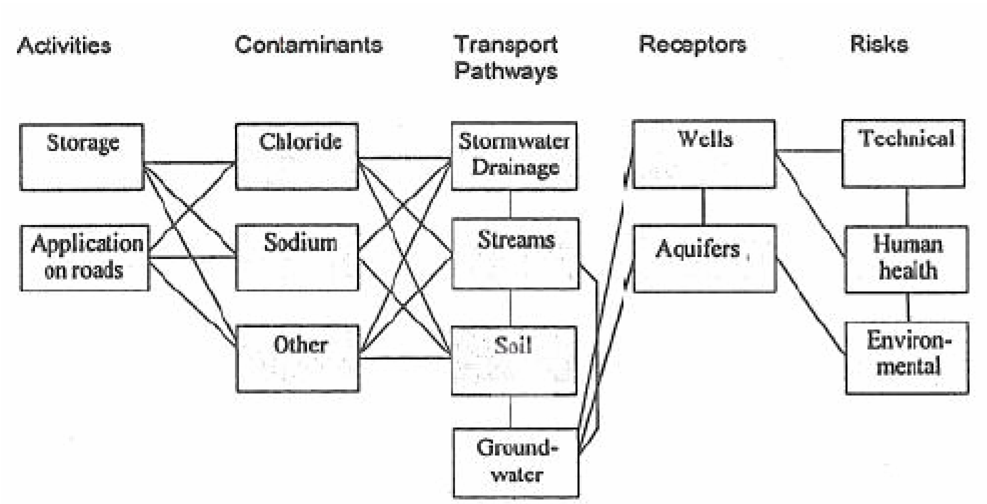

Figure 5: Possible sequence of events for identification of risk factors from aquifers and groundwater wells from de-icing of roads (Rosén 1998).

10.6 Physical-based models for the dispersal of de-icing chemicals in soil and groundwater

Flow in soil and groundwater is based on Darcy's law, or a modification of this, in order to describe the hydraulic conductivity as a function of water content in the unsaturated zone ($K(\theta)$). Modeling of flow in the unsaturated zone is generally more complicated than in ground water because the water pipeline capacity in the soil depends on water content, this means that the speed of the pollution and water depends on how much water the soil contains. A more detailed description of the flow equations included in models for saturated and unsaturated zone is shown, for example in Bear and Verruijt (1992), and the user manuals for the different models are mentioned later. In the following sections an overview will be given of the simpler models that can be used to calculate a rough estimate of the transport speed, the dispersal and dilution.

To get a rough estimate of vertical velocity in the unsaturated zone, v, one can use equation 5 (Appelo and Postma, 1996) which assumes stationary conditions, i.e., infiltration and water content is constant over time:

$$v = \frac{P}{n_w} \qquad \textbf{(Equation 5)}$$

where P is the precipitation surplus (m/year) (that which infiltrates and is transported further down, i.e., precipitation - evapotranspiration), and n_w is the water filled pore volume. Estimates based on this equation are very approximate; the assumption of stationary conditions is virtually never the case in the unsaturated zone. In soils with low infiltration capacity the precipitation will in many cases not infiltrate but form the surface runoff so that it becomes more difficult to find a good estimate of P, this is discussed more in a later paragraph.

In the saturated zone the assumption of stationary conditions is more realistic, particularly in some larger systems (alluvial deposits, larger eskers). In such systems there may be small fluctuations because of the water levels in relation to the extent of the aquifer. Appelo and Postma (1996) describe a simple box model for flow in a homogeneous open groundwater area (Figure 6). The method is based on the known ground water supply (P) and distance to the groundwater separation, as well as the effective pore volume (n_e, the active part of pore volume the water flows through.

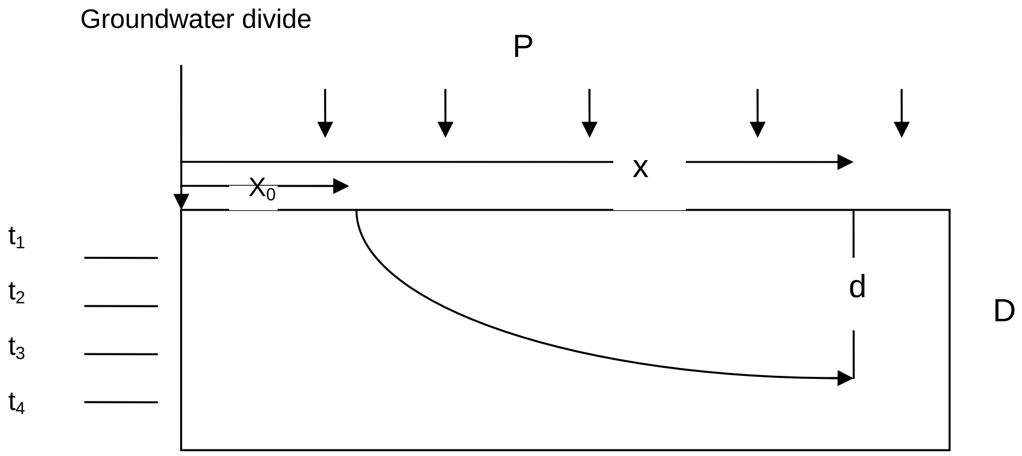

Figure 6: A vertical cross section through a homogeneous groundwater area. One can, for simplicity's sake imagine that this is a schematic representation of the system in Figure 2 and that the road is at X_0.

For the system in Figure 6, the following equations describe the velocity v_w at x:

$$v_w = \frac{dx}{dt} = \frac{Px}{n_e} \qquad \textbf{Equation 6}$$

dx/dt distance per unit of time, ne is effective pore volume (i.e. the part of the pore volume taking active part in transportation, this is always less or equal to the total pore volume). Alternatively, we assume a groundwater system where the flow is considered one-dimensional and use Darcy's equation; we must know the groundwater slope (dh/dx), the saturated hydraulic conductivity in aquifer (Ks) and the effective porosity in the area we want to calculate the pore water speed:

$$v_w = \frac{-K_s \dfrac{dh}{dx}}{n_e} \qquad \textbf{Equation 7}$$

To calculate the concentration C*(x, z, t)* at a given time (*t*) in point (*x, z*), one can use different analytical equations, which pore water speed (v_w) is known and we consider a vertical section through an aquifer. A modified form of Domenico and Robbins' equation shown in Domenico and Shwartz (1998) provides:

$$C(x,z,t) = C_0\left(\frac{1}{4}\right) erfc\left[\frac{R_f x - v_w t}{2(\alpha_x v_w t R_f)^{1/2}}\right] erfc\left[\frac{z}{2(\alpha_z x)^{1/2}}\right] \qquad \textbf{Equation 8}$$

where C_0 is the supply concentration, *t* is time, αx and αz are dispersal coefficients respectively *x* and *z* directions, v_w is pore water speed (which can be calculated from equations 6 or 7 or by use of a more advanced flow model), R_f is the retardation factor, *erfc* (the mathematical expression of "complimentary error function ").

If the substance we are interested in adsorbs, as is the case for the cation part of deicing agents, the retardation factor Rf can be calculated from:

$$R_f = 1 + \left(\frac{1-n}{n}\right)\rho_s K_d \qquad \textbf{Equation 9}$$

where K_d is the distribution coefficient and ρ_s is the density of the soil, *n* is porosity.

Another simple analytical solution to calculate the concentration CNA, Cl in a certain well is described in Åstebøl et al. (1996):

$$C_{Na,Cl} = \frac{1}{3.15 \cdot 10^7\, T(dh/dl) + LN)} M 10^3 \qquad \textbf{Equation 10}$$

where M is amount of salt per year (kg/m), T is transmissivity (m2/s), dh/dl is the hydraulic gradient, L is distance between the road and well (m), N is the groundwater input per year (m/year) i.e., N-E, as described in previous equations.

10.6.1 Advanced numerical models

If one would like to map the flow around specific sensitive areas, e.g. drinking water wells and water works, it may be necessary to make more accurate studies in which one takes into account the geological structure and groundwater flow. Then it may be appropriate to use numerical models in which one can describe factors that affect ground water, both the natural factors and the measures that can be implemented to prevent negative consequences (removal of surface water from the road, the manipulation of groundwater flow by use of pumping wells etc.). We have not found examples in the literature on the use of pumping wells to prevent the supply of salt from the road to the wells, but this is the methodology used in relation to other point pollution. We have not included examples of it here. Internationally there are many models to choose between, both commercial and free programs. Some of these are briefly described in this section, and references to web addresses for the different models are listed in Table 8. Models which are designed only to describe what happens in the top portion of the soil, often with the plant growth, nutrient capture, etc. are often one-dimensional, i.e., they describe only a vertical column, and all transport and turnover occurs in this direction. Examples of 1 D models are: Coup (Jansson & Moon, 2001) and SWAP (Van Dam et al. 1997; and Kroes et al. 2008). Coup is developed in Sweden and it therefore includes freezing and thawing processes together with snow accumulation and melting. Lundmark (2008) used this model to describe the amount of salt (mg m^{-2}) at different distances from the road. Out of a desire to look at the dispersal in the road/roadside area the system a two-dimensional model would be better. A two-dimensional model study using Hydrus (described below) showed that the water that infiltrated along the road, can infiltrate into the road surface, this may be helping to erode materials located in the foundations of the road (Apul et al., 2007), which could potentially provide additional pollution loads to groundwater.

Examples of models that take water and element transport in both two and three-dimensional unsaturated and saturated systems are SUTRA_2D3D (Voss and Provost, 2003), Hydrus (http://www.pc- progress.cz / Fr_Hydrus.htm), Feflow (http://www.wasy.de/english/produkte/feflow/index.html). With Hydrus it is the standard to be able to model multiple pollutants simultaneously, for example, both salt and Potassium Formate, but you can not model the density flow, i.e. flow caused by the water for example with high salt content sinks below the water with a lower density. SUTRA can model this density flow, but only one pollution component at a time, for example, either Na + or Cl - or Formate. A widely used model for groundwater flow is Modflow (http://www.swstechnology.com/software_product.php?ID=12) which models both water and element transport (several components at once) one of the most commonly used user interfaces are VisualModflow. There are many examples of this is used for both Norwegian and foreign groundwater systems. It is also the model used as a management tool to maintain basic water balance at Oslo Airport Gardermoen (Jarl Øvstedal pers.comm.). Most of the models mentioned above have not included the effect of frost that can lead to reduced water pipeline capacity (K) in soil and greater runoff on the surface.

Table 8: URLs to the models mentioned in the text

Modell	Webadresse
Coup	http://www.lwr.kth.se/vara%20datorprogram/CoupModel/index.htm
SWAP	http://www.swap.alterra.nl/
SUTRA2D-3D	http://water.usgs.gov/nrp/gwsoftware/sutra.html
Hydrus	http://www.pc-progress.cz/Fr_Hydrus.htm
Visual Modlfow	http://www.swstechnology.com/software_product.php?ID=12
FeFlow	http://www.wasy.de/english/produkte/feflow/index.html

10.7 Data requirements for model simulations - uncertainty

Before you start with modeling, it is important to know what you want answers to, something which is essential when selecting the correct model. This includes, among other things, a specification of the relevant resolution in time and space; For example, is a calculation of actual concentrations in a drinking water well that is located 50 m from the road relevant? Or the effects on the source output area a few miles downstream from the road? Or is it in regards to the flora and fauna of the road community, or adjoining agricultural land? Or, does one wish to present danger of undesirable salt (deicing agents) exposure along the road network in Norway?

As shown by the equations above, it is necessary to have good data on climatic conditions, infiltration (precipitation, evapotranspiration, surface runoff and groundwater input) and model parameters (hydraulic conductivity, dispersal constants, porosity, coefficient, and decomposition constants) which describe the physical composition of the road surface / environment. If a national sensitivity map in relation to the salting it will be of great importance for certainty in the model calculations that good data on infiltration in various areas is used, this will be a major

challenge because the meteorological data collected represents only a relatively small range. Therefore, this must be considered in future work (set up the desired resolution in time and space). What time resolution required will depend on how big the system you want to map. In a study conducted by Rutter and Thompson (1986) of salt dispersal from the road, it is recommended to use day values for supplies of water and salt. For the system which Eliasson (2000) studied day values were used, but concluded that the leveling effect of a powerful unsaturated zone would make using weekly values sufficient, this would make the simulation time shorter. In Sweden, where several of the studies in this literature study refers to, weather data from SMHI and VViS, which is the Swedish road-weather information service, is used according to Lundmark (2003). They have about 600 stations every half hour measuring precipitation, wind (speed and direction), and temperature in the air and on the road.
As described earlier, it is essential for the fate of deicing agents after they have left the road, how much water it can be diluted with, and in the case of groundwater, how much water there is infiltrated. There are different methods for calculating the precipitation surplus (i.e. precipitation - evapotranspiration), and a widely used model for determining evapotranspiration is Penmans equation (Penman, 1948) or a more complex model such as the Coup (Jansson and Moon, 2001), these will, based on the input of precipitation, air temperature, global radiation, wind, plants (and soil type, only for the Coup) estimate the precipitation surplus. Coup may also provide an estimate of the distribution of excess water on surface runoff, ditch runoff and groundwater feeding. There are also other similar models to calculate the local water balance, but only a few include the processes of freezing, thawing and snow accumulation/melting.

For the choice of physical parameters the foundation of the map which includes topography, Quaternary geology, groundwater, vegetation and land use be important. If one is modeling the effect of different de-icing strategies in relation to a specific distance, it is probably necessary to perform local measurements of both physical parameters and depending on the proximity of a meteorological station, possibly make additional measurements locally, and collect data that can be used to validate the model. Again, we look at what's available of this type of data in Sweden. Lundmark (2003) used the land use map based on the EU's CORINE Land Cover project, with a resolution of 1-25 Ha. Digital soil species maps are found with 50 x 50 m resolution, and hydro-geological map covers virtually the entire Sweden, and there are digital elevation models with a resolution of 50 x 50 m.

10.8 Examples of models used

Lundmark (2008) calculated the fluctuations in chloride concentrations in the upper 0.2 m of soil profile at different distances from the road (1,5 and 10m) by combining the dispersion model of Blomqvist (1999, see equation 2) and the Coup model. Results of this modeling shows concentrations ranging between 0-300 g Cl^-/m^2 (1m from the road), 0-80 g Cl^-/m^2 (5m from the road) and 0-15 g Cl^-/m^2 (10m from the road), these results are comparable with measurements. The studies were conducted at Kista, ca. 10 km north of Stockholm, along the E4 with an annual usage of sodium chloride at ca. 16 tons/km.

To calculate the combination of vertical flow in unsaturated zone and horizontal transport in groundwater zone, Lundmark (2008) used a combination of the Coup model and a simple drainage model (Hooghoudt, 1940), as well as hydrogeological maps for the area to determine the direction of groundwater flow. The various calculations were integrated into a GIS tool such that the potential chloride concentrations found at different distances from the road could be estimated. The estimates for Kista (described above) showed maximum concentrations at ca.600 mg Cl/l. The model concept developed by Lundmark (2008) is an improvement of the model developed by (Gontier and Olofsson, 2003); because the latter does not take into account the direction of groundwater flow. The model concept developed by Lundmark (2008) is intended to be used to map the risk areas of high salt concentration along the Swedish road network. The studies by Lundmark (2008) showed that soil type had the greatest impact on salt concentrations in the root zone, while the vegetation had the greatest impact on water balance. The highest concentrations in the soil were modeled just before snowmelt, and the lowest concentrations were estimated in October/November after dilution with autumn rainfall. This is in accordance with field observations and simulations of the transport of de-icing chemicals at Gardermoen (French et al. 2002)

Howard and Maier (2007) illustrate the effect of the development of urban areas on the sodium chloride concentration of groundwater near Lake Ontario. There is a number of underlying groundwater wells in the area. Visual Modflow was used in these calculations, and only the upper aquifer (open aquifer) results were summarized. Salt use varies between 20-250 tons of sodium chloride per. km road (from dual carriageways to multiple lane motorways). Long-term simulations show that according to the reference situation, one can expect a stabilization of the salt concentration in the upper aquifer after approx. 700 years, and a maximum concentration in the range of 5,000 mg of sodium chloride/l along the highway with the highest burden. A development increased area with impacted groundwater (maximum concentrations slightly above 200 mg sodium chloride / l), and a stabilization of the concentration after ca. 100 years. It is important to note that these are very site-specific calculations that may not in any way be transferred to Norwegian conditions. Generally the interesting part about this study is that it illustrates how long it may take before concentration stabilizes, and that you get an example of how such a model can be used in relation to specific sites with high sensitivity, for example in connection with drinking water supply, vulnerable habitats along the source horizons, etc. Granlund and Nystén (1998) also used the transport module that exists in Visual Modlfow, MOC to simulate salt dispersal in Finland. In the most recent study, the model was also integrated with GIS tools.

Åhnberg and Knecht (1996) also performed modeling with SUTRA in addition to the simple mass balance calculations described earlier. They stress that this type of tool is important in relation to the testing of hypotheses about the system and increased process understanding.

Niemi (1998) modeled chloride concentrations in different types of aquifer in Finland, different scenarios and sizes aquifers was simulated, the results showed that the size of aquifer was of greatest importance.

In a model study from Nybroåsen in Sweden (Eliasson, 2000) concludes that the development of the E22 can give chloride values greater than the Swedish drinking water norm of 100 mg/l at some of the local wells in extreme precipitation episodes. The study calculates the residence time of salt from the road to some of the wells for 35 days in such individual episodes. Eliasson (2000) used the model FeFlow in their studies. This study concludes that heavy metals stored in the soil from the existing road can be washed down to the groundwater due to increased salt supply.

To model the transport and decomposition of organic de-icing chemicals at Gardermoen SUTRA was used (French et al., 2000, 2001). In this case, a Monte-Carlo procedure was used to obtain the uncertainty in transport speeds due to variability in the soil. Flesjø (2007) also used SUTRA to model salt transport in the saturated and unsaturated zone along Rv35 as a result of various infiltration conditions. The model was also used to test whether salt transport was affected by density effects.

Brommeland (2006) used Visual Modflow to calculate the opportunity to put down an extra drinking water well at Kroksjøen water utilities in Eidskog. Here the potential impact of salt from Rv2 was also a problem, which is followed up further in a master's thesis at UMB (estimated completion in 2008).

Ostendorf et al. (2006) found a good match between modeled and observed salt concentrations in groundwater around a salt storage facility.

10.9 Measures

Measures to reduce the negative consequences of de-icing chemicals are described in separate chapter; here some are described briefly with a special focus on the protection of groundwater. To prevent an unfortunate impact on groundwater while maintaining friction requirements on the road, different measures may be appropriate. Of course provided that the mechanical removal of snow and ice are optimized the need for chemical de-icing is as little as possible:

- Prevent infiltration along the vulnerable stretches by collecting all surface runoff from the road (see Chapter 12)
- Select biodegradable de-icing chemicals (Chapter 5.2) and possibly optimize the degradation of these through fertilization with nitrogen and phosphorus.
- Choosing a combination of different de-icing chemicals such as urea and organic de-icing chemicals to achieve fertilization effect as mentioned in point 2. There are no concrete examples of this being used elsewhere.
- Manipulating groundwater flow so that vulnerable recipients or installations (wells) do not receive drainage from the road. This is done in connection with contaminated sites, but we have not found references where this is done along the road.

10.10 Assessment of methodology in relation to Norwegian conditions

In this section we have pointed out some factors that should be taken into account if you want to take advantage of the model strategies described in the preceding paragraphs for Norwegian conditions.

When it comes to transferring the dispersal equations discussed in Section 10.3 to Norwegian conditions parameters should used with caution and only as an indicator of a possible dispersal pattern. Location specific constants will depend on the amount of salt used and wind conditions along the road.

Mass balance calculations such as those mentioned under Chapter 10.4, will be useful for a regional assessment of the most impacted areas. In this way, areas where it is necessary to take measures or areas where it may be necessary for more detailed studies are easily identified (for example precipitation, evapotranspiration and soil conditions) and can be mapped. To use these regional computational tools, it is important to have local data for various meteorological parameters precipitation, temperature, wind, global radiation, etc. Potential evapotranspiration is in some cases calculated for a single measurement station, but this is not always the case, so that in equation 4 this variable will be more uncertain than the actual rainfall. According to the Meteorological Institute Homepage (met.no) there are only 223 operational weather stations in Norway, there are in addition, weather stations in connection with NVE and Bioforsk's measurement network, and the Norwegian Public Road Administrations own weather stations.

NVE has a monitoring network consisting of about 64 measurement areas for monitoring groundwater condition (a total of 81 measuring points). At 55 of these areas additional ground water temperatures are measured.

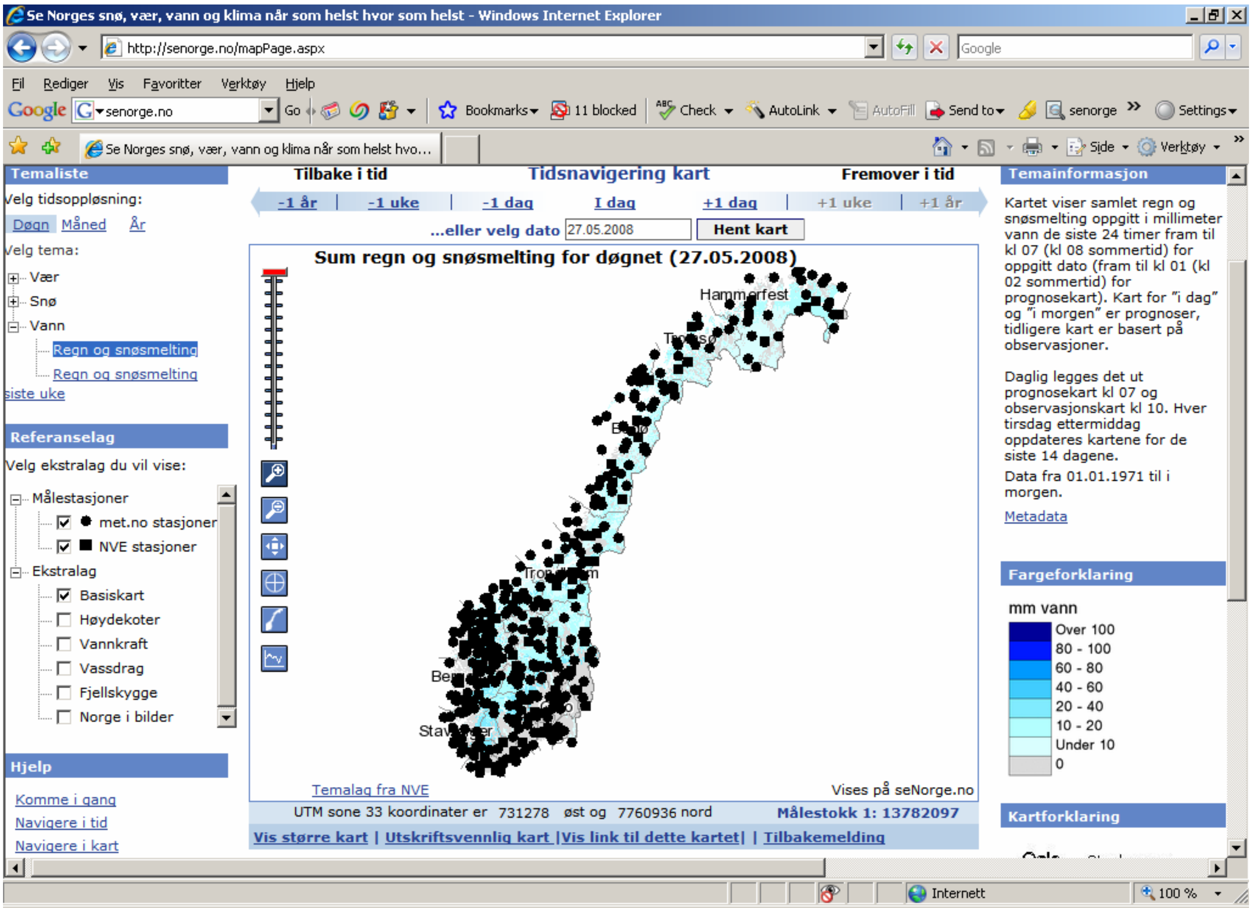

Figure 7: Screenshot from seNorge.no showing monitoring stations in Norway that are the basis for calculating of rain and melting snow on a square kilometer level for the entire country.

At 15 sites soil water parameters are monitored (soil temperature, soil moisture and ground frost depth) these are run by Bioforsk. On the website seNorge.no there is also daily information on snowfall, rainfall, temperature and available water (combination of melting snow and rainfall). In Figure 7, the distribution of monitoring stations from NVE and met.no is shown. On the basis of this the weather situation for each square kilometer in Norway is calculated.

Where there are no local meteorological stations nearby it will be necessary to estimate values for these areas. There are different downscaling methods to make such calculations that take into account local topography, elevation etc. (see, for example, Srinivasa, 2006). Such methods are not reviewed here. Calculations of evapotranspiration are done to a limited extent, but are an extremely important variable and take into account how much water is actually contributed to groundwater recharging and is thus crucial for the dilution effect. Drainage situation, such as ice on the frozen ground, will also affect how and how much water and de-icing chemical infiltrates a specific area.

In addition to the salt coming from the road, it is important to take into account the natural salt supply present due to proximity to the ocean or natural salt sources in the local sediments.

Statistical tools (chapter 10.5) can not be used in the predictive context. It is designed for one type of area (geology and natural conditions). It is therefore not possible to transfer the classification system directly to other localities. Another weakness of the methodology is that the salt quantities and climatic factors are not included in the risk assessment. Presented in another study are the measurements of chemical parameters from 13000 wells in Sweden (Olofsson and Andström, 1998), Norway has also registered many wells (Granada database, see www.ngu.no), but because there is a more complex groundwater system, with water in the mountains and water in the soil and large topographical differences, it is possible that there are not enough wells to perform a similar analysis in Norway.

When it comes to choice of alternative de-icing chemicals, those mentioned here are mobile and degradable, and can therefore be a convenient alternative to salt near vulnerable groundwater resources. Depth of the groundwater must be adequate for the chemicals degrade before they reach the groundwater something which depends on the amount of infiltration and concentration. A number of models can be used to calculate the critical load (maximum load along a stretch of road) under certain conditions, such as geological conditions, topography, and water balance and flow direction. Some examples of problems where detailed modeling may be appropriate:

- A waterworks is located near the road; they want to increase the water outlet, where they can put an additional well?
- A landowner has the high salt concentrations in their wells, how likely is it that this stems from the road?
- Can the groundwater flow be changed locally to prevent the addition of salt from the road?
- Which concentration of organic de-icing chemicals can be broken down before they reach groundwater?

Dilution is the most important measure to reduce concentrations of road salt. To calculate the dilution potential for the national road network, various mass balance models are used together with information about salt supplies and excess rainfall (infiltration). Various risk tools are developed to make such calculations; risk tools based on geological and topographical conditions are also developed. More detailed models that can simulate water transport, dilution and degradation, will be useful where you have particularly valuable groundwater resources. These will require good meteorological data and data on ground conditions.

By all type of modeling, national or local, it is important to take into account the uncertainty in both input values and parameters that describe the soil conditions. In such an assessment a sensitivity analysis is included. This means that the input values or physical parameters are systematically changed to see what effect this has on the results. By making such an analysis we can determine which data are most important in order to map better. The usual procedure of modeling is that the model is calibrated against measured data. It can later be validated against independent data, either from another field, or by testing model to a later or earlier time-series than the model is calibrated for. If the model also works well in a different field location where you have measurements to compare, one can use the model in unmeasured field locations. Also it will be necessary to make an uncertainty analysis; one can do this either by Fuzzy Logic method. That is, using a combination of more and lesser-known extreme values of physical parameters (minimum and maximum values) and guide values or by Monte Carlo simulations. The latter method entails running the model with a large number of realizations of the basic conditions (for example, different variations of the Ks values, porosity, etc.) in addition to that you can test the effect of uncertainty in input values (input variables such as climate conditions).

11. Chemical and biological effects of de-icing chemicals on surface water

11.1 Changes in water quality over time as a result of road salting

Ramakrishna and Viraraghavan (2005) highlights the three main points of chemical changes in lakes: 1) Changes in density gradient, 2) Increased chloride concentration, and 3) Changes the circulation pattern due to increased salt concentrations. A whole series of articles have been published that demonstrate the relationship between increased salt concentrations in lakes and road salting. Large regional surveys in the United States, where the chemical composition of thousands of lakes have been studied, show a clear correlation between the density of buildings and chloride concentrations in lakes (Langen and Prutzman 2006, Mattson et al. 1992; Munson and Gherini 1993). Several sites with flowing surface water (streams and rivers) in northern Europe and in The United States have also shown an increasing trend in chloride concentration over the past decades, and the primary explanatory variable is mainly road salt (Albret in 2005, Godwin et al. 2003; Goldman and Lubnow 2000; Koller 2008; Nedjai and Rover 2001; Peters and Turk 1981; Ramakrishna and Viraraghvan 2005, Roseberry et al. 1999; Ruth 2003, Scott 1981). Large areas of North America have, therefore, challenges related to surface water quality regarding aquatic organisms and drinking water supplies (Howard and Maier 2007). In England (Windemere - English Lake District) road salting has increased the concentration of both sodium and chloride up to 100 times in some streams that naturally only receive low concentrations of sea salts through precipitation and soil (Sutcliffe and Carrick 1983a, b).

In relation to the number of publications related to salinity problems in lakes in the United States, the number of published articles from Nordic lakes and rivers is more modest. However a number of Nordic studies have been published. Wike (2006) looked at the effects of road salt (sodium chloride) on water chemistry at Skånetjern pond in Gardermoen. The concentration of chloride has increased significantly over the past 35 years due to the application of sodium chloride, but no changes in the horizontal gradients or permanent separation of the water layers has been detected. Thunqvist (2003a, 2004), who wrote a doctoral dissertation on topic (Thunqvist 2003b), has conducted significant studies on the long-term effects of road salt on the environment in Sweden. Thunqvist has also worked out models for chloride concentrations in surface water (Thunqvist 2000; 2003b). The model is simple and accounts for the sodium chloride added via road salting from different ranges and dilution gradient infused via precipitation (minus evaporation). The model does not account for concentration of chloride in precipitation, which can vary significantly with distance from the sea, but appears to work well in the areas it has been tested and where the chloride from the road is the single largest source.

Bækken and Haugen (2006) studied 59 lakes in Norway based on a data material of about 1200 lakes. This is the largest study of the influence of road salt regarding water quality in Norwegian lakes as of today. They selected 59 lakes were less than 200 meters from road salt, and a similar number of control lakes near these were used as references. It was documented that 18 of the 59 salt affected lakes had clearly developed salt gradients (the difference between water from the surface layer and bottom layer> 10 mg Cl/l), and 17 of these had also clearly developed oxygen gradients in the autumn with lower oxygen concentrations in bottom water than in the top echelon (differences of> 6 mg O^2/l between bottom water and water from the top echelon). They also analyzed a number of other metals (Fe, Mn, Cu, Cd, Zn, Pt, Cr, Rh, Ni and Ca), as well as the polyaromatic hydrocarbons (PAHs) in the water phase. The differences between the quality of water from the upper layer in the group with salt affected lakes versus the control group showed as expected significant differences regarding the concentration of sodium and chloride. The study found especially large differences in Østlandet, which perhaps can be attributed to the selected reference lakes that appear to be somewhat further from the sea than the road salt affected lakes.

With the chloride concentrations measured in the surface waters in the salt affected lakes (<40 mg Cl/l) (Bækken and Haugen 2006) sea salts with precipitation can also be contributing to the significant increase in concentrations of sodium and chloride in relation to the reference lakes. In the lower water section the salt affected lakes lack of oxygen also contributes to higher concentrations of iron and manganese in the water phase. Average daily traffic (ADT) also related well to the amount of salt in the lakes. Initially there is nothing to imply a direct correlation between ADT and salt runoff, but ADT can probably be set in relation to the amount of salt used on roads, which in turn can be related to increased runoff of salt to streams and lakes.

In Norway, the best known examples that deal with road salt (primarily sodium chloride) and lake stability are probably associated with Padderudvann and Svinesjøen, meromictic lakes near Asker (Bækken and Færøvik 2004; Færøvik et al. 2005; Kjensmo 1997). Padderudvann has in recent decades received a powerful stability increase due to road salt application. Svinesjøen has in contrast shown a reduced stability in the water masses. In the case of Svinesjøen this decrease in stability is because the upper parts of the heavy bottom water layer (monomolimnion) in Svinesjøen has a higher content of salts from road salting, so that the stabilizing element is reduced. A literature review of the phenomenon of stability and meromixis in lakes can be found in the doctoral degree to Hongve (2004). Hongve (2004) describes various circulation patterns and how various parameters such as wind (kinetic energy), surface area and total depth (stabilizing factor; gravity point) affects the circulation pattern. Each lake will have its morphometric peculiarities, which in turn will affect the circulation pattern.

11.2 Effect of sodium chloride on the circulation of surface water

If a lake is to maintain good oxygen conditions and stable nutrient content in water, vertical circulation is important. Salt concentrations in the lake affect the stability conditions (for example, Rimmer et al. 2005), and modified density gradients and layer stability have in a longer period been pointed out to be a potential consequence of the use of road salt (Gibbson and Stewart 1972; Judd 1970; van de Voorde et al. 1973). If a lake receives added salts from the outside, this will potentially lead to a more permanent chemocline (chemical layering) in the lake (Goldman and Lubnow 1992). Hakala (2004) has studied meromixis in relation to several processes, and mentions specifically four cases that lead to meromixis. 1) Supply of saline water. 2) The supply of nutritious/turbid water. 3) The supply of groundwater. 4) Mitigating circulation opportunities because of the lake morphology and/or precipitation topography. Additions from road salting fall mainly under category 1, but will have elements of the other categories also.
Streams and rivers will not develop chemical layers such as a lake. They will also have a significant rapid change in water quality, related to road salt (for example, Demers and Sage 1990), but salt concentrations dilute downstream the source of salt due to the addition of more salt-poor waters from other parts of the catchment area. Larger rivers will generally have lower concentrations and relatively less variation in chloride concentrations due to a greater dilution effect of applied salt.

11.3 Effects of sodium chloride on the heavy metal concentration in water

Bækken and Haugen (2006) generally found significantly smaller differences between road salt affected lakes and control lakes regarding the concentration of PAHs and other metals than sodium. Löfgren (2001) studied the effect of sodium chloride in the stream water of five catchments in southern Sweden, and he found, however, that applied sea-salt as a deicing agent led to significant cation exchange reactions in the catchment area, whereas sodium largely replaced calcium and magnesium in the soil. In the same way as natural sea-salt episodes can lead to acidification episodes by ion exchange with hydrogen and aluminum in acidic soils (e.g. Hindar et al. 1995), or increased concentrations of metals and base cations such as calcium, potassium and magnesium from the soil with higher base saturation, the use of sodium chloride or sea-salt as a deicing chemical does the same thing. Löfgren (2001) recorded also cation exchange between sodium and hydrogen ions, zinc and cadmium. This resulted in, amongst other things, increased concentrations of zinc and cadmium in some of the lakes. Significant correlations between the measured concentrations of sodium chloride and zinc in Scandinavian surface waters have also been reported by others (Ruth 2003). Mason et al. (1999) showed that the use of sodium chloride resulted in increased concentrations of calcium, potassium and magnesium, in addition to Na and Cl. They also found the highest concentrations of base cations both during snowmelt and during the autumn rain floods. This suggests that sodium chloride from road salt has apparently remained in the system in this area over long periods, such as there's enough time to create equilibrium between the exchangeable cations in the soil and Na from road salt (Mason et al. 1999). Sodium is retained in the catchment area to a greater extent than chloride via cation exchange in the catchment area (for example, Rhodes et al. 2001), and can be washed out several months after the last road salting (Cherkauer 1975). Enrichment of road salt in the soil has also been shown to attribute to increasingly higher concentrations in the river and stream water by accumulation, probably due to an increased saturation of salts in the soil (Kelly et al. 2008; Werner and Dipretoro 2006).

11.4 Effects of sodium chloride on the organic material in water

A relatively new problem, as opposed to increased chloride concentration and chemical layers in lakes which has not been particularly discussed in the literature regarding road salting and water quality, are changes to concentrations of natural organic matter (humus) in relation to application of chloride to a catchment area. Correlations have recently been demonstrated between sea-salt episodes and the humus concentration in water (Haaland 2008; Monteith et al. 2007). Higher chloride concentrations lead to higher ionic strength in the soil solution and reduce the solubility of humus substances. The waters become clearer. This is analogous to what has been observed in connection with the acidification process in the Norwegian surface waters in recent decades. Humus flocculates and the lake will lose an important buffer system. Lake light and temperature conditions will also change. At present we know little about what role de-icing chemicals can play in this context.

11.5 Concentrations of sodium chloride in surface waters

Runoff from road salt deposits in the United States have been shown to contain very high concentrations of chloride, and concentrations of more than 10 000 mg Cl/l have been measured (Evans and Frick 2001). These are extreme chloride concentrations in a freshwater context. By comparison, the chloride concentration in Bottenviken is about 3000 mg/l, and the ocean contains, on average, approximately 19 000 mg Cl/l. Such high concentrations have been measured very locally in streams and rivers in connection with snow melt episodes. In several rivers in The United States has the chloride concentrations been more than 200 mg/l and it is often reported to be due to road salting (for example, Heath 2004). Extreme concentrations in streams and rivers have been measured in conjunction with high precipitation and high runoff of melting snow in the winter, and chloride concentrations over 1000 mg / l have been measured (Erickson and Arnason 2004; Evans and Frick 2001). This is either at locations near the source of the salt (concentrated road salt in snow) or somewhat further away, and usually only over shorter periods of hours to a few days. By comparison, the chloride concentration in Norwegian lakes mainly is in the range 1 to 10 mg/l. Surface water nearby the coast, however, has naturally a higher content of chloride due to sea salts in precipitation, but concentrations of chloride are rarely than 30 mg/l. Extreme sea salt episodes in Norway from winter 1993 stand out (Andersen 2002), and high chloride and sodium concentrations in streams and rivers were recorded several months after the episode. The concentrations of chloride are still nowhere near those that have been measured locally near road salt storage sites (cf. Evans and Frick 2001) and remains essentially well below 50 mg/l.

Chloride concentrations in lakes in Norway and in other countries, are generally explained by the distance to the sea and if the locality drains marine sediments or not - whether the locality is above or below marine limit (for example, Smart et al. 2001). The use of sodium chloride, calcium and Magnesium, sea salt and others as road salt, has modified this picture considerably for a number of affected localities (Bækken and Haugen 2006; Granato et al. 2004; Jackson and Jobbàgy 2005, Kaushal et al. 2005; Ramstack et al. 2004, Siegel and Livermore 1984; Siver et al. 1996). Salt gradients have been found in small artificial lakes where shopping centres and asphalt make up large parts of the catchment area and large amounts of sodium chloride and calcium chloride is used as deicing agents (Cherkauer and Ostenso 1976). This is a simple point, which underlines the fact that some aquatic sites can be influenced by road salting to a greater degree unlike others. Direct runoff to lakes that located just next to the road will be a problem. Distances up to 100 meters or more are likely to significantly vary in response and consequences are more uncertain as a result of road salt runoff.

11.6 Sodium chloride and the effects on aquatic flora and fauna

11.6.1 General

Biological effects as a result of salting may be measured in several ways. Direct measurements in the laboratory where "Key organisms" are exposed to different concentrations of salt (salt gradients) is one way to determine the effect-levels, observations in the field is another more time consuming, but often a more ecologically relevant way to determine the effects.

Regarding acute effects, these are generally measured in tests lasting less than 4 days (96 hours). The most common time frames for acute tests is 24 hours, 48 hours, 72 hours and 96 hours. One usually wants use results from tests done over the longest possible time, so that tests with duration of 96 hours are often preferred when the PNEC (predicted no effect concentration) levels to be established. The results from acute tests are relevant to use when short-term effects of salting is to be evaluated (snowmelt, runoff periods etc).

Testing of the chronic effects takes place over a longer period of time (more than 5 days). These tests are more relevant and necessary when long-term effects of salt in surface water are to be evaluated.

A potential major stress for aquatic ecosystems in flowing surface water (streams and rivers) will be pulses related to the high runoff in the winter (Ramakrishna and Viraraghavan 2005). Exposure time is important here to assess the dose-response relationship. At a long exposure time lower concentrations will cause damage.

11.6.2 Acute effects

Sanzo and Hecnar (2006) conducted acute (96-hours) and chronic (90 days) laboratory tests with tadpoles of the forest frog (Rana sylvatica) which are among the most common amphibians in North America. The acute experiments were conducted with concentrations from 0 to 9750 mg/l, while in the chronic experiments concentrations varied from 0 to 1030 mg/l. This concentration range was also measured in surface water in the area before the experiments started (Sanzo and Hecnar 2006). The acute effects occurred in a concentration range of 2636-5109 mg/l. Mortality and abnormal behavior occurred relatively quickly (<24 hours). Most of the tadpole in both the acute and chronic tests responded to the salt by displaying both physical and behavioral changes. The results from the chronic tests showed that increasing salt concentration (0-1030 mg sodium chloride/l) decreased survival, resulted in reduced activity and weight of the tadpole, as well as increasing the extent of physical changes. Sanzo and Hecnar (2006) concluded that sodium chloride had toxic effects on tadpole of forest frogs at concentrations found in water located nearby a salted road.

Kefford et al. (2004) compared test systems for measuring the salt tolerance of invertebrates in freshwater. Among the organisms tested were the limpet (Burnupia stenochorias), shrimp (Cardin nilotica), mayfly (Euthraulus elegans) and water beetle (Micronecta piccanina). There were sometimes large differences in sensitivity between these species. Water beetle had LC_{50} values of approximately 1000 mg/l, while the shrimp endure far more salt (sodium chloride) i.e. approximately 15,000 mg/l). LC_{50} values for limpet and mayfly were ca. 6000 mg/l and 7000 mg/l. The results of the project showed that there were no significant differences in test results between the use of stagnant water and running water in test systems, as well as that the use of pure sodium chloride in the tests resulted in larger effects than the use of artificial sea-salt (which contains more of other salts, including calcium and magnesium).

A number of results in the references above are summarized in Table 9.

Table 9*: Overview of the effect concentrations for aquatic organisms (see text above.*

Name of organism	Scientific name	Duration	LC_{50} (mg sodium chloride/l)	Reference
Forest Frog	Rana sylvatica	96 hours	2636-5109	Sanzo and Hecnar (2006)
		90 days		
Limpet	Burnupia stenochorias	96 hours	~ 6000	Kefford et al. (2004)
Shrimp	Cardin nilotica	96 hours	~ 15,000	Kefford et al. (2004)
Mayfly	Euthraulus elegans	96 hours	~ 7000	Kefford et al. (2004)
Water beetle	Micronecta piccanina	72 hours	~ 1000	Kefford et al. (2004)
Caddisfly	Limnephilidae	96 hours	3526	Blasius and Merritt (2002)
Crustaceans	Gammarus	96 hours	7700	Blasius and Merritt (2002)
Mayfly	Hexagenia Limbata	96 hours	2400	Blasius and Merritt (2002)
	Hexagenia Limbata	96 hours	6300	Blasius and Merritt (2002)
	Tricorytus sp.	96 hours	2200-4500	Blasius and Merritt (2002)
	Lepidostoma sp.	96 hours	6000	Blasius and Merritt (2002)
	Callibaetis fluctuans	96 hours	> 5000	Benbow and Merritt (2004)
Crustaceans	Chaoborus americanus	96 hours	> 10000	Benbow and Merritt (2004)
	Hyallela aztec	96 hours	> 10000	Benbow and Merritt (2004)
Snail	Physella integra	96 hours	> 10000	Benbow and Merritt (2004)
Fish, rainbow trout	Oncorhynchus mykiss	96 hours	20,380	Vosylienė et al. (2006)

Blasius and Merritt (2002) point out that there are a wide variety of aquatic insects that are well adapted to life in brackish and saltwater. Blasius and Merritt (2002) have also completely a relatively comprehensive review of effect studies on macroinvertebrates (most of them in the laboratory) with sodium chloride in running water. Salt doses of 10000 mg sodium chloride/l over a period of 96 hours have generally shown little or no effect on macroinvertebrates in streams, but exceptions exist (see Table 9).

Effect studies on the fry of rainbow trout (Oncorhynchus mykiss) gave LC_{50} values of 20380 mg sodium chloride/l (Vosylienė et al. 2006). Effects on some blood parameters were also measured and showed that concentrations of chloride of 180 mg/l led to a reduced number of red blood cells and increased content of hematocrit in the blood, but had no significant effects on, for example, the hemoglobin content (Vosylienė et al. 2006). Although the relationship between the number of red blood cells and state of health of fish is not clear, the results suggest that physiological changes takes place at much lower concentrations than those that result in death.

Benbow and Merritt (2004) came largely to the same conclusions at the completion of laboratory experiments and microcosm experiments in fields: effect levels (LC50, 96 hours) on the mayflies (Callibaetis fluctuans, Chaoborus americanus, Physella integrators) and the snail (Hyallela aztec) were all in the concentration area of 5000 mg sodium chloride/l. Also chronic tests (15 days) showed that these organisms can tolerate large amounts of salt. Compare these effect-levels with average values for the content of chloride in surface water in Michigan (18-2700 mg Cl/l, median 128 mg Cl/l), there is little suggesting that the concentrations in the water near a salted road will pose a threat to these organisms.

11.6.3 Acute effects on relevant Norwegian species

In general, it appears that over a relatively short period of time, it will take high concentrations of sodium chloride for the dose to be fatal (Benbow and Merritt 2004). Most species of aquatic fauna have LC_{50}values of well over 2000 mg sodium chloride/l with duration from 1 to 4 days (Table 10). Experiments in Canada with fish species similar to ones present in Norway, indicates critical levels of over 500 mg Cl/l with more than a week of exposure (Evans and Frick 2001). Salmonids (trout and rainbow trout) appear to be more sensitive than perch fish and koi carp, but there is likely to be considerable variation in tolerance within the various fish families. Most freshwater species generally have a high tolerance for chloride and seem to tolerate concentrations far above what is common to find in most rivers and lakes in Norway. Plankton and macroinvertebrates seem to be somewhat less tolerant of high chloride concentrations compared to fish (Evans and Frick 2003, Evans 2004, Table 10). Freshwater plant tolerance is often between 250 - 1000 mg Cl/l (Evans and Frick 2001; Evans 2004). Generally the transition from holomixic to meromixic has been proven by many to be harmful and/or lethal to benthic invertebrates (Hauser 2004). Few studies have looked at the effects of de-icing chemicals on phytoplankton, especially for species adapted to ionically poor lakes the effect concentrations (EC_{50} - the concentration that gives 50% reduction in growth response) of chloride on such Norwegian phytoplankton species, shows in part a considerable variation, with 5 mg Cl/l non-marine diatom Aulacoseira distans to more than 5000 mg Cl/l for green algae Selenastrum capricornutum (Færøvik 2006). To avoid damage to more than 90% of the phytoplankton species the chloride concentration must not exceed 25 mg/l (Færøvik 2006).

Table 10: Effect Concentrations (fatal, LC_{50}) for aquatic fauna and flora relevant for Norway. Varying exposure time. Fauna: 1 - 4 days. Flora (plant plankton): 5 days. Data on laboratory animals in newly fertilized stages from egg until adult specimens. Fish up to juvenile (young fish) stage. Eels in both glass-eel and eel wandering stage. Standard deviations for the experiments are indicated. Data from the PAN Pesticide database (http://www.pesticideinfo.org/).

Name of organism	Scientific name	LC_{50} mg sodium chloride/l	LC_{50} standard deviation	No. of trials	Category
Sectional Worms					
Leech	Erpobdella sp .*	8000	1000	5	Not acutely toxic
Earthworm	Limnodrilus hoffmeisteri	6381	595	8	Not acutely toxic
Earthworm	Nais variabilis	2569	-	1	Not acutely toxic
Crustaceans					
Woodlice	Asellus sp .*	7095	1731	11	Not acutely toxic
Fish					
Eel	Anguilla rostrata *	19665	1785	2	Not acutely toxic
Goldfish	Carassius auratus	8170	1218	53	Not acutely toxic
Crucian carp	Carassius carassius	13750	-	1	Not acutely toxic
Rainbow trout	Oncorhynchus mykiss	6778	684	4	Not acutely toxic
Insects					
Spring midge larvae	Cricotopus trifasciatus	6221	-	1	Not acutely toxic
Mosquito larvae	Culex sp.	10350	150	2	Not acutely toxic
Caddisfly larve	Hydropsyche sp.	9000	-	1	Not acutely toxic
Mussels					
Heart mussel	Cerastoderma edule **	66000	-	1	Not acutely toxic
Lung Snail	Lymnaea sp.	3400	12	2	Not acutely toxic
Nematodes					
Nematodes	Caenorhabditis elegans *	21721	4096	9	Not acutely toxic
Phytoplankton					
Diatoms	Nitzschia linearis	2430	-	1	Not acutely toxic
Zooplankton					
Rotifer	Brachionus calyciflorus	3664	-	1	Not acutely toxic
Water flea	Daphnia magna	4879	1166	16	Not acutely toxic
Water flea	Daphnia pulex	2260	790	2	Not acutely toxic

* The species used in the experiment are not with us in Norway, but we have similar and closely related species.
** High value (outlier) in the group, i.e. that this value may be too high to represent other closely related species.

11.6.4 Long-term effects (chronic effects)

Concentration changes of sodium chloride occur rapidly in urban streams and rivers (Ruth 2003), and are often correlated with changes in drift (benthos in river which release themselves in to the water masses to escape adverse conditions) (Evans and Frick 2001). A correlation between increased chloride concentration in streams and rivers and changes in the macroinvertebrate community has also been found (Evans and Frick 2001). Increased chloride concentration appears to reduce the species composition, but there are also examples that some species that are able to survive and increase in number (Novak and Bode 2004). Over time, it is probable that a lower number of salt tolerant species will out-compete the more salt intolerant species, analogous to how it is the estuaries (areas where fresh water meets sea water), where we often find a reduction in species composition the closer we get to the sea which is the same as high salinity in the river (Bulgar et al. 1993). Lakes that become meromictic have also been shown to have changed the structure of the aquatic ecosystem (Judd et al. 2005). Smoll et al. (1983) saw by using microfossil analysis of diatoms and chrysophyta in sediment from a road salt impacted and a now meromicitic lake in the United States, a modified species composition and a greater proportion of species that thrive better in nutrient-poor lakes. Both oxygen deprivation and increased concentrations of nutrients and salts make the monimolimnion to an unsuitable habitat for most plants and animals, and only specially adapted species will survive in this layer.

For lakes that in addition to salt supplies receive nutrients from sewage and agriculture, it may be difficult to assess the effect of road salting on fauna and flora. Road salt will initially lead to less nutrient-rich waters in the mixolimnion, while sewage and runoff from agriculture will lead to eutrophication. One such example is at Padderudvann close to Asker (cf. Bækken and Færøvik 2004).

Horrigan et al. (2005) used multivariate statistical methods (such as "Artificial neural networks" ANN) to estimate the tolerance or sensitivity of different groups of invertebrates to the conductivity in water and to create a salinity index that links changes in salinity to changes in the composition of invertebrates. The basis for doing this was that there were extensive records of many various invertebrates (50-60 for two different habitats). The results of this work showed relatively low values for conductivity (0.8-1.0 mS/cm or approximately 1000 mg sodium chloride/l) changes in the macroinverebrate society could be detected i.e. that more salt-tolerant species take over with increasing salt concentrations in water. It is important to be aware that this work is carried out in Australia, and does not deal with changes in salinity due to road salting, but due to natural changes in salt levels (which is a common problem in Australia).

Buckler and Granato (1999) show in a literature review of 15 studies where biological effects of deicing agents have been investigated. They refer to, among other things, studies in which chronic effects (increased migration of benthic organisms) were observed at concentrations of chloride above 1000 mg/l. In a field experiment (also referred to by Buckler and Granato 1999), where sodium chloride (1000 mg/l) was added to a stream the occurrence and diversity of algae decreased, while the bacterial density increased as a result of the reduced quantity of predators.

11.7 Summary: acute and chronic effects

Environment Canada (2001) summarizes the results of the investigations that are referenced above (acute and chronic) as well as results from a number of other studies (Figure 8).

Figure 8: Experimental data from acute tests (acute tests <4 days = dark symbols, acute tests 1week = gray symbols (middle) and predicted chronic toxicity (open circles on the left).

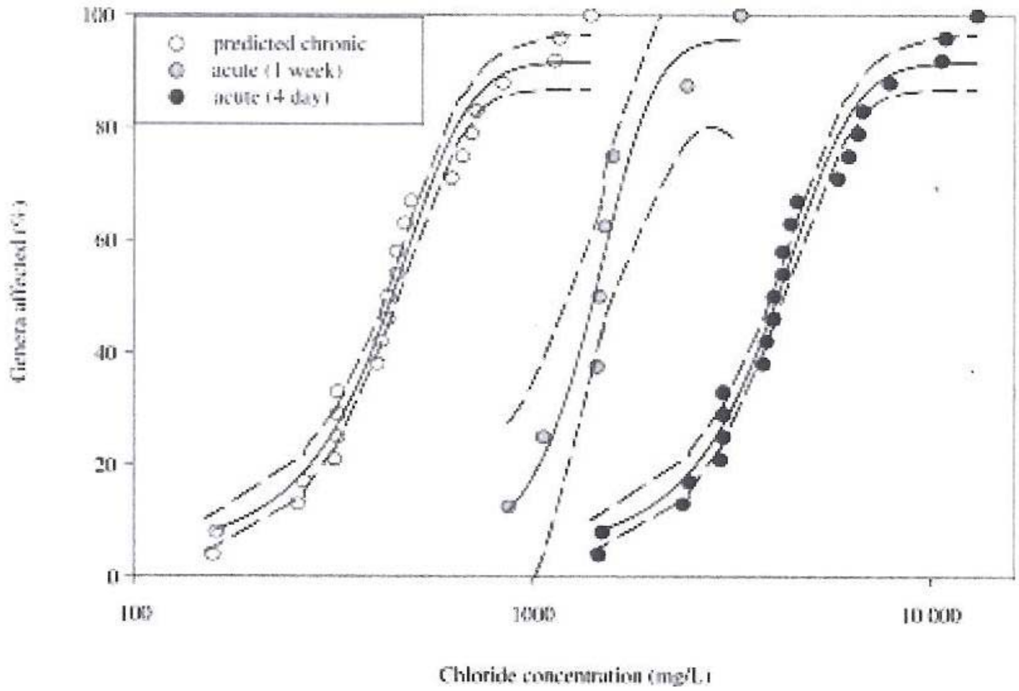

Figure 8 shows that the acute effects (exposure <4 days), acute effects of exposure to 1 week and chronic effects occur at the Cl-concentrations, respectively (approximately) 6000 mg/l, 1100 mg/l and 560 mg/l if we allow the effect on 50% of the organisms. If you were to allow the effect on only 5% of organisms' chronic effects would occur after a few days at a Cl-concentration in the range from 200 to 250 mg/l (U.S. EPA 1988, Environment Canada 2001).

U.S. EPA (1988) developed the following water quality criteria for chloride:

- 4-day average of chloride (when it was associated with sodium) should not exceed 230 mg/l more than 1 time every 3 years on average.
- 1-hour average of chloride should not exceed 860 mg/l more than 1 time every 3 years

U.S. EPA mention in this context that these levels do not provide complete protection if the chloride is associated with K, Mg or Ca. Evans and Frick (2001) found that potassium chloride and Magnesium chloride was more toxic than sodium chloride. Fish appear to be less sensitive to calcium chloride than to sodium chloride, while the opposite is apparently true for invertebrates.

11.8 Effects of other de-icing chemicals

Fischel (2001) has done a fairly thorough review of the literature to compare the chloride based (sodium chloride, calcium chloride and Magnesium chloride) and acetate based (potassium acetate, sodium acetate, calcium magnesium acetate and a mixture of potassium acetate and calcium magnesium acetate) de-icing chemicals. The comparison was made with respect to several parameters, including effects on water quality and aquatic organisms.

Fischel (2001) ranked de-icing chemicals with respect to toxicity to trout, water fleas and aquatic plants (Table 11). The general trend is that the chloride based de-icing chemicals are less toxic to these aquatic organisms than those based on acetate. Sodium chloride is the least toxic substance among the chloride based, while calcium magnesium acetate is the least toxic among the acetate based chemicals. Ice Ban + Magnesium chloride and potassium acetate is based on the results of the four test organisms in the water, the most toxic de-icing chemicals.

Table 11: Ranking of de-icing chemicals with respect to toxicity to trout, water flies, and algae. Data from Fischel (2001).

	Acute toxicity Rainbow trout (Oncorhynchus mykiss)	Acute toxicity Water flea (Ceriodaphnia)	Chronic toxicity Water flea (Ceriodaphnia)	Growth Inhibition Algae (Selenastrum)
1 (least toxic)	NaCl (23%)	NaCl (23%)	NaCl (23%)	NaCl (23%)
2	CMA (25%)	$MgCl^2$ + Caliber	$CaCl^2$	$MgCl^2$
3	$CaCl^2$	CMA (25%)	$MgCl^2$ + Caliber	$CaCl^2$
4	CMAK	$CaCl^2$	$MgCl^2$	Ice Ban + $MgCl^2$ (50:50)
5	$MgCl^2$ + Caliber	$MgCl^2$	CMA (25%)	CMA (25%)
6	$MgCl^2$	NaAc	CMAK	$MgCl^2$ + Caliber
7	NaAc	CMAK	KAC	CMAK
8	KAc	KAc	Ice Ban + $MgCl^2$ (50:50)	KAc
9 (most toxic)	Ice Ban + $MgCl^2$ (50:50)	Ice Ban + $MgCl^2$ (50:50)		

NaCl=sodium chloride, $CaCl_2$=calcium chloride; $MgCl_2$=Magnesium chloride, $MgCl_2$ + Caliber = Magnesium chloride + carbohydrate from corn; CMA = calcium magnesium acetate; CMAK = calcium magnesium acetate + potassium acetate; KAc = potassium acetate; NaAc = Sodium acetate; Ice Ban + $MgCl_2$ = by-product from agriculture (carbohydrate) + Magnesium

The relatively clear trends in the toxicity of chloride based and acetate based de-icing chemicals summarized by Fischel (2001) are confirmed by studies reported by Joutti et al. (2003). Here the effect of sodium chloride, calcium chloride, Magnesium chloride, potassium formate, potassium acetate and calcium magnesium acetate on the root-lengthening of the onion (Allum cepa), growth inhibition of duckweed (Lemna), enzyme activity (“reverse electron transport test”, RET) and the effect on bacteria (Vibrio fischeri) (i.e. other organisms than those Fischel (2001) emphasized in his literature review). Plants (onion and duckweed) were the most sensitive organisms, and had the lowest EC_{50} values for the organic deicing agents, while RET was more sensitive to chloride salts. In general, deicing agents were highly toxic or toxic to the organisms tested. Take in consideration that that the organic relations (formate, acetate) are broken down in the unsaturated zone, however, it is less likely that effects will occur in water (Joutti et al. 2003). This will naturally depend on the organic deicing agents infiltrating the soil and not flowing directly into surface water.

Some studies have shown that calcium magnesium acetate can be a good alternative to sodium chloride in regards to the effects on phytoplankton (e.g. Goldman and Lubnow 1992). They found no negative biological effects with the use of calcium magnesium acetate. Calcium magnesium acetate can probably still have a negative consequence in that the degradation of acetate consumes oxygen, so that the use at certain locations will result in oxygen deprivation (Albright 2005).

11.9 Freshwater species' critical loads to sodium chloride

Critical load is an estimate of how much nature can receive a pollutant without incurring damage (Larssen and Høgåsen 2003). In Norway the critical limits for the acidification from strong acids (sulfur and nitric acid) to surface waters and forest soils and eutrophication of terrestrial vegetation with nitrogen has been developed.

For surface water a critical load is set to maintain a self-reproducing trout population.

If we think in these terms for road salt, the relationship between salt, levels of salt and different effects must be known and the acute, chronic and possibly sub-lethal effects must be distinguished between. Based on the international comparison of toxicity data of chloride such relationships seem to be relatively well known (see above) and critical loads for fish and invertebrates can probably be established today by visiting in depth the data that already exists.

It must, however, be discussed which organisms and habitats are the most sensitive and how much protection is desirable (and economically advisable).

There is considerably less data available regarding the effects of other de-icing chemicals than sodium chloride and critical loads for these will be associated with greater uncertainty.

11.10 Discussion and conclusions

An assessment of how the application of elevated concentrations of sodium chloride affects aquatic flora and fauna has been made. In studies it has been documented that the toxicity of runoff from the road is higher in the winter months, parallel with the increased application of road salt. Increased toxicity could not only be explained with concentrations of chloride, and it is also due to mobilization, increased bioavailability and increased toxicity of other traffic caused pollution components. EPA has established limits for the chronic toxicity of chloride of 230 mg Cl/l and acute toxicity of 860 mg Cl/l. In practice, fish could tolerate short chloride exposure up to approx. 6000 mg Cl/l.

In an overall assessment road salt creates greater environmental problems in lakes associated with chloride and increased toxicity of other pollutants components in road salt. The risk of environmental effects of chloride and mobilized pollution components are considered to increase related to surface water systems that provide accumulation of road salt. The environmental effects can be reduced by reducing and adapting the use of road salt together with the establishment of surface water systems that do not accumulate road salt with a risk of leaching in toxic concentrations.

Several species of fish, including pike (Esox lucius), perch (Perca fluviatilis) and others, can be fished in estuary zones. This probably requires an adaptation over time for each species. Episodes with high concentrations of salts therefore can probably have an effect on organisms that have adapted to high salt concentrations. Freshwater species are adapted to coastal areas are also likely to tolerate the effects road salting better. The smolting process of anadromous salmonid, which takes place in the spring, will probably also overlap in time with the snowmelt period of some rivers catchment area. Salmonid in the smolting stage are essentially more sensitive to changes in water chemistry than in salmon in the pairing stage, referred to in Table 9 (cf. i.e. Berntssen et al. 1997).

In general, it seems that the critical loads (based on acute effects) for aquatic organisms are very high. Concentrations of more than 2000 mg sodium chloride/l are shown in experiments to be typical values for what organisms can tolerate before they begin to die out (see Table 7 and 8). Concentrations appear to be high compared to those usually found in surface water in Norway. Effect concentrations can be significantly lower. Rainfall along the coast may in some cases contain concentrations of chloride up to 30 mg/l, and areas that drain marine clays will also have naturally higher concentrations of chloride than in upland areas. Although these are low concentrations in relation to what has been measured as a salt impact during the snow melt in some streams and rivers, generally it is probable that there will be fewer effects on flora and fauna of the coastal lakes compared to lakes that have a naturally low salt concentration. Evans and Frick (2001) conducted a literature review for surface water in Canada and found few or no effects of sodium chloride in coastal streams and rivers. Fauna in small streams where concentrations and doses can locally be very high would still probably be the most vulnerable. Generally phytoplankton is probably not as vulnerable in acidification periods due to the low primary production during snowmelt periods in Norway. A general increase in salt concentration in the lakes will probably be of greater importance (Færøvik 2006).

12. Measures against the effects of de-icing chemicals and surface water management

12.1 Road salting and salt concentrations in run-off water

The quantity of salt used on the Norwegian road network has increased dramatically in recent years. In the last 8 years the amount of salt used on the Norwegian road network has more than doubled (Figure 9).

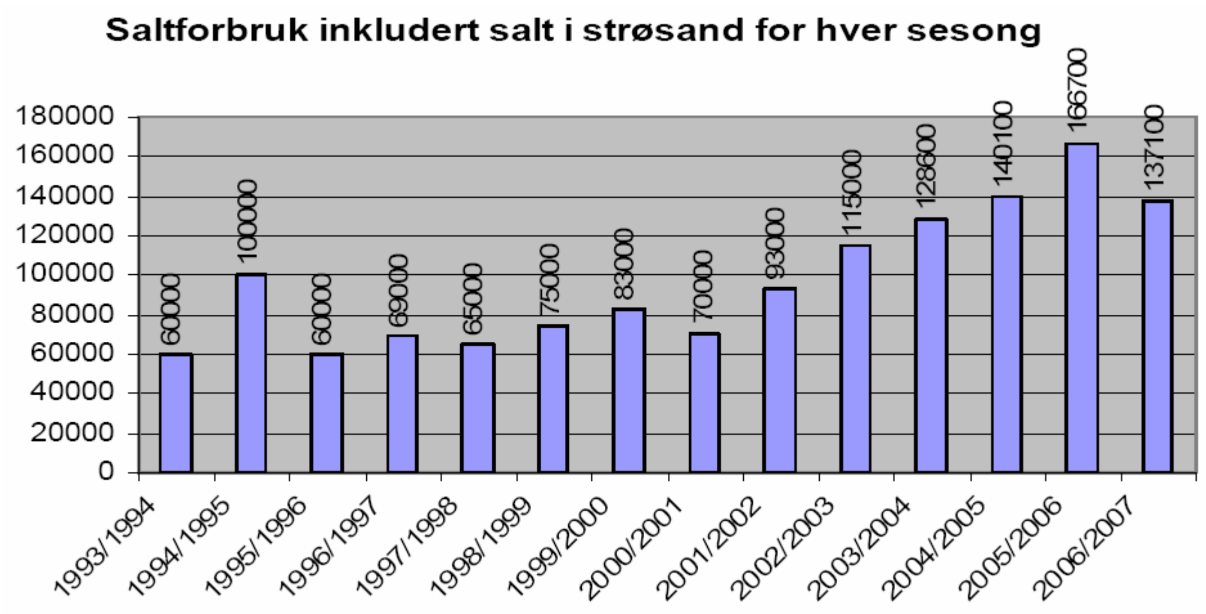

Figure 9: Shows the consumption of road salt each season from 1993/94 until 2006/07 (Sivertsen 2008).

Increased salt consumption will lead to an increase in salt quantities that are transported in run-off water from the road. Measurements were made in the period 1998 to 2007 have periodically shown high concentrations of chloride in run-off water from the road (Røhr and Åstebøl 2002, Åstebøl and Coward 2005, Amundsen and Roseth 2004, Bækken et al. 2005 and Roseth 2006). The measurements have shown that normal concentrations of chloride in run-off water is in the range of 50-1000 mg Cl/l, while high concentrations may be in the range of 3000-5000 mg Cl/l.

The need for measures to reduce leakage from road de-icing chemicals should be considered from an overall compilation of vulnerability and the possible effects on the recipients and groundwater along the new road. The assessment should include rivers downstream the road up to a point where the dilution is considered to be so good that no negative effects can be expected. Since road salt, and particularly chloride, is a conservative element that is difficult to remove, the measures must comply with the directing of run-off to less vulnerable areas or dilution to concentrations that do not provide potentially damaging effects.

12.2 Current run-off water management

New highways are always constructed with a storm water system. Normally this consists of a water system of open ditches (often grass-covered) in the middle part and on either side of the road. Run-off from the road is collected in ditches and led to a soakaway with an internal distance 60-100 m. In the road ditches it is possible that infiltration, evaporation and sedimentation occur depending on the design and drainage. From the soakaway and sand traps the run-off water is collected into main pipeline drainage. This drainage pipe directs the water out to a low point for release or to treatment for run-off water.

Modern road constructions in Norway and internationally, always include systems for treatment and disposal of surface water (Marsalek 2003 and Åstebøl 2006). In Norway, the measures relating to run-off water management are largely focused on metals, oil and organic pollutants in the water, and less on road salt. Recent efforts to purify and balance the surface water in Norway include (Åstebøl 2006):

- Stormwater pond/sedimentation pond
- Wetlands
- Sand filter/purifying filter
- Vegetative methods

Of these, stormwater ponds especially, are used in the construction of new highways in Norway, but there are also examples of the use of infiltration in grassy depressions in areas with permeable soils and water-filled gullies along the road. In relation to the construction of the new E6 Gardermoen – Biri, infiltration of surface water in the adapted ditches along the road. The intention is to be able to reduce waste water from roads and to significantly reduce the amount of run-off that reaches a watercourse in a flood situation. In areas where high-traffic roads cross groundwater resources of regional importance for water supply, a complete sealing of the storm water system is needed to get run-off with road salt directed out of the vulnerable area. Such measures are implemented where E6 crosses groundwater at Sannom which is used as the water supply for Lillehammer (Norway).

In the following figures and photographs various systems for run-off management used in Norway are shown (Figure 10-15). When it comes to stormwater ponds, the most commonly used management technique, there is great variation in shape and hydraulic function. When designing soakaways, both water depth and transfer depth for treated run-off water may cause differences in hydraulic function.

Figure 10: Stormwater pond for treatment of surface water E6 at Taraldrud in Ski municipality Norway) with integral soakaway and water depth in the main pool of 2.5 m (Photo: Roger Roseth).

Figure 11*:* **Stormwater pond at E6 Skullerud in Oslo (Norway) with an integral and closed soakaway and water depth of 1.5 m (From Åstebøl and Coward 2005).**

Figure 12*:* **Stormwater pond at E6 in Råde (Norway) which consists of a separate sedimentary basin with hard bottom followed by a shallow wetland filter (0.5 m) with overflow to the stream (Photo: Roger Roseth).**

Figure 13*:* **Water-filled gully with local thresholds/dams for infiltration and sedimentation of the added surface water.**

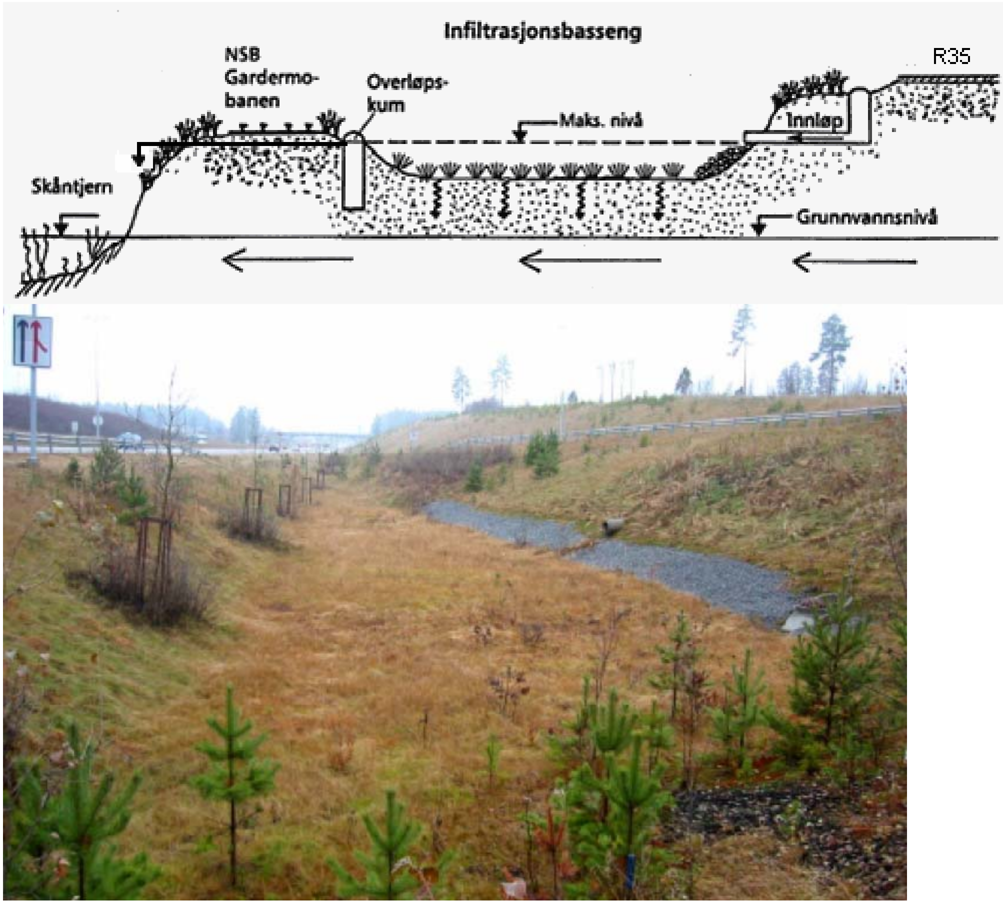

Figure 14: Grassed surface water channel for the infiltration of surface water at a junction on the E6 towards Oslo Airport (Norway)(From Åstebøl 2006).

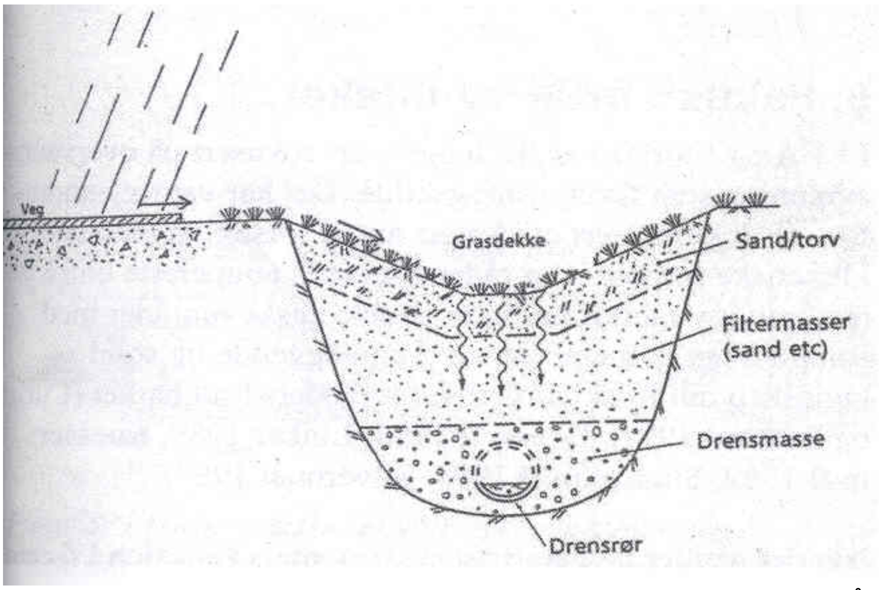

Figure 15: Figure 15: Principal sketch of contruction of a soakaway (from Åstebøl 2006).

Heavily trafficked roads through urban areas represent very serious problems associated with management of pollution and road salt in run-off water. High degree of sealed areas along the road means that the majority of traffic created and other urban-made pollution is washed to the run-off water by rain and snow melting. In urban areas of great recreational value there is little land available to build treatment systems and in many cases it is watercourses which receive run-off water. Åstebøl (2007) has given a review and presentation of the compact treatment systems that may be appropriate to use in urban areas. The report does not focus on road salt, but considers the cleansing of other pollution components in road run-off. In this report the treatment system described are based on pretreatment for storing and removal of particles followed by various purification solutions. Also for these treatment systems it will be necessary to assess the effects of road salt and salt layering for purification processes and whether the proposed treatment systems might contribute to the affect or even equalize salt pulses added by selective melting out from snow.

12.3 Measures against salt pollution

There are no simple and cost effective treatment methods that can remove road salt (sodium chloride) in the run-off water from the road before it reaches the groundwater, streams or lakes. Removing chloride is especially difficult, which is a conservative element commonly used as a tracer to study the flow of water. During transport through the soil sodium will be removed or retained through ion exchange processes, but the process can cause the mobilization of other metals which are environmentally damaging.

Below is a list of measures that can help to prevent adverse effects of road salt:

- **Dilution** - hydro-technical measures which contribute to the dilution of applied road salt in the soil, groundwater, terrain, large watercourses or in stormwater ponds

- **Accumulation and drainage** - hydro-technical measures to bring the salty run-off water from stretches of road with run-off or infiltration to the vulnerable and valuable recipients (groundwater, valuable streams and lakes that are vulnerable to salt) and direct this run-off to less vulnerable areas (large watercourses, discharges to the sea or infiltration in less vulnerable areas).

- **Reduced use of road salt** – the applied quantity of road salt used will be reduced through the optimization of procedures for dispersal, required quantity, optimization of spreading equipment, increased use of snow ploughs at less snowfall etc.

- **Removal and disposal of salt-rich ploughed snow** - in cities space is limited for local storage of cleared snow. Parts of the salt-rich ploughed snow will be driven to landfills which should be constructed so that the snow melts in to the less vulnerable recipients. For motorways outside built-up areas can consider milling snow in to areas with increased distance from the road and road ditches to increase local infiltration.

- **Alternative de-icing agents** – consider the use of alternative deicing agents on stretches of road with vulnerable recipients. The alternative de-icing agents have different characteristics in relation to run-off and aquatic environments, and which agent to be used should be considered based on local conditions.

- **Technical treatment solutions** - there are a number of technical treatment methods developed for desalination of seawater, of which the most common are distillation, reverse osmosis, electro dialysis, and membrane distillation. On the basis of known technologies are these methods assessed as inappropriate for desalination of large quantities of surface water from the road. This is due to the overall costs of energy consumption, investments and maintenance. Commercially available treatment system for drinking water based on reverse osmosis or distillation can be used to remove salt from a local drinking water supply.

- **Controlled drainage of saline** bottom water - in ponds and lakes along heavily trafficked roads there has been a proven salt layer, with potentially permanent oxygen-free conditions in bottom water. Such effects can probably be prevented by arranging for a controlled removal of the saline bottom water, which in some cases can be performed without the use of energy at a siphon principle.

Of the aforementioned measures, there are some which are applied or are planned to be applied in Norway.

Controlled dilution of surface water with road salt can occur through infiltration along the edge of the road, i.e. that it is adapted for roadside infiltration, ditches and surface water channels along the road. Application of this management measure requires local soil with adequate water conductivity and that the road is not located over a vulnerable groundwater resource or with vulnerable vegetation downstream of the road.

The construction of the new E6 at Biri (Norway) (National Road Administration 2007) describes a solution where the infiltration first goes through filter masses, for improved treatment of other traffic created pollution components (Figure 16). The infiltration trench is constructed so that there is a backwater and increased infiltration before the water goes in the overflow into the storm water system which leads the water for discharge at sag-curve. By extreme run-off, which exceeds the hydraulic capacity of storm water system, surplus of water is led to an open ditch. Herein it may be considered whether it might be possible to skip a conventional storm water system in such a solution.

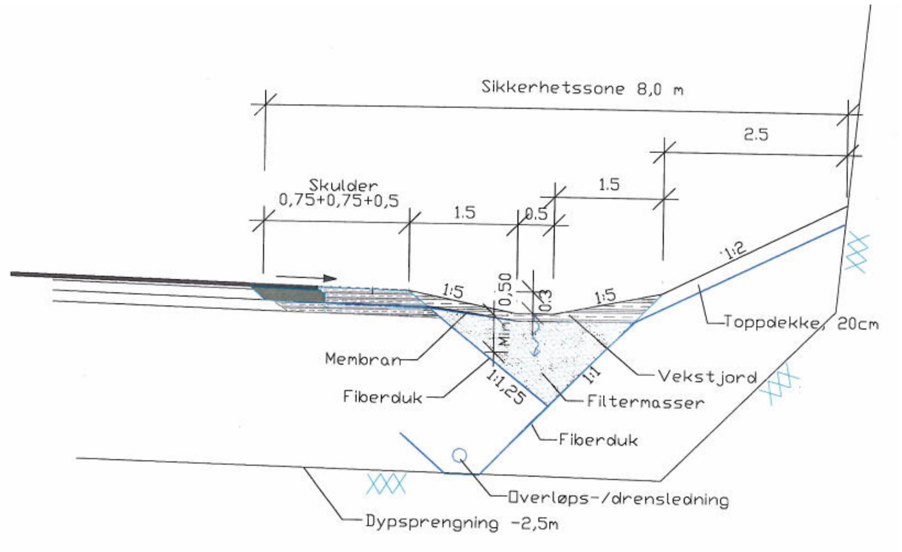

Figure 16: Sketch of the infiltration trench at E6 Gardermoen - Biri (Norway) (Norwegian Public Roads Administration 2007).

For the protection of valuable groundwater resources used for drinking water have efforts been made in small areas to divert run-off water with road salt to less vulnerable recipients.

Such measures are carried out to protect groundwater used as drinking water for Lillehammer (Norway) on the E6 at Sannom and for a similar area along the E4 in Sweden (Figure 17). The solutions are different in that one is based on the sealing of the road ditches, while the other collects salty water that flows from the road through a combination of curbs and drains.

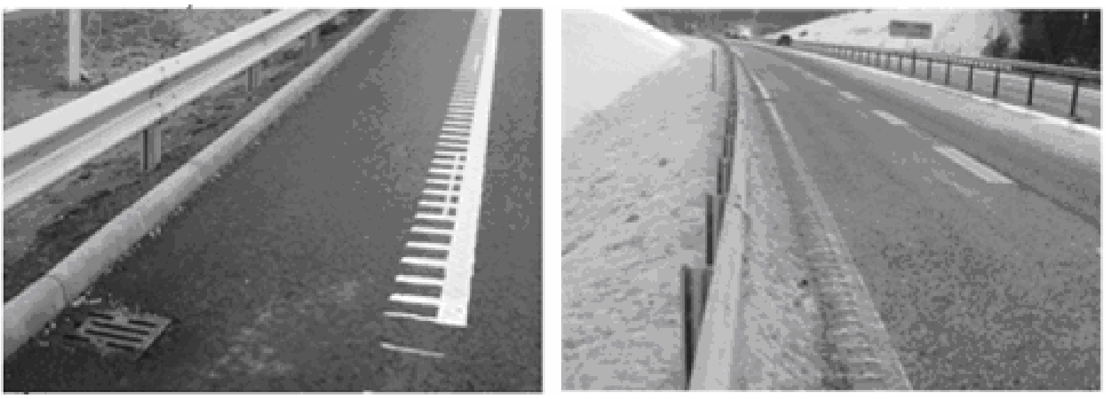

Figure 17: Collection of saline runoff from the road through the gullies and drains on E4 at Begraåsen in Sweden for the protection of groundwater resources used for drinking water (Lundmark, 2008).

12.4 Technical treatment systems

A range of technical treatment systems have been developed for the desalination of sea water to drinking water. The methods are developed to desalt water as cheaply and efficiently as possible. Miller (2003) developed an overview of technical systems used to remove the salt. The most important and most common solutions are also described by Habimana (2006). In the following paragraphs a brief description of the most appropriate technical solutions for desalination and removal of sodium and chloride in water is given (12.4.1-12.4.4).

12.4.1 Distilling

Salt water boils and the water vapor condenses. Energy needs are estimated at 620 kWh per m3 of fresh water produced. Alternatively, the water is heated to 100 ° C under pressure and is directed to a tank without pressure where a rapid expansion of the water occurs with the formation of water vapor that condenses into fresh water. This method is called multiple stage flash evaporation (MSF), and is less energy demanding than conventional distillation, approximately 103.5 kWh per m3 of fresh water.

12.4.2 Reverse osmosis

Salt water is directed through a membrane under high pressure. Salt will be filtered out. The energy consumption will vary with salt concentration in the water, and is estimated to range from 4.9 kWh per m3 of water formed from seawater. In the runoff, and groundwater that contains road salt the salt concentration will be somewhat lower something that will affect the energy consumption associated with reverse osmosis. Reverse osmosis is commercially available in small filter solutions adapted to purification of drinking water for scattered settlements.

12.4.3 Electrodialysis

Electrodialysis is the transportation of salt ions through an ion-exchange membrane to another solution under the influence of an applied electric potential difference. The energy consumption will vary with salinity in the water. At low salt concentration the energy needed is 3.8 kWh per m3 of fresh water.

12.4.4 Membrane distillation

Water vapor is forced against a porous hydrophobic membrane which only allows the passage of water vapor. Freshwater formed by condensation on the other side of the membrane can not flow back to the salt water, since the membrane is impermeable to water. The method has low production capacity of fresh water, and often considered as having low potential because of this.

In an overall assessment of the current methods reverse osmosis is recommended as the most appropriate for the production of fresh water from salt water. Smaller commercial devices based on reverse osmosis or distillation is available for use on private drinking water supply.

12.5 Stormwater ponds and salt

Stormwater ponds or run-off water/sedimentation basins are now almost a standard solution for cleaning and balancing of surface water from large newly built highways in Norway (Åstebøl 2006). Studies have shown that stormwater ponds can provide an effective removal of many traffic caused pollution substances such as heavy metals, PAHs, and oil (Åstebøl and Coward 2005, Bækken et al. 2005 and Semandeni-Davies 2005). This is especially the case with components of pollution attached to particles that can be sedimented out in to the ponds.

If the water from the road is to be treated in a stormwater pond it is usually a wish that most of water should be directed in to the pond. It is also desirable that the stormwater ponds treat run-off water from such a long stretch of road as possible.

Intuitively such a drainage strategy results in potentially adverse effects of road salt in vulnerable recipients, but this is minimally studied and there is no certain evidence that there is an adverse effects in vulnerable recipients as a result of such conditions. Intuitively, the drainage of run-off water could direct a larger proportion of the dispersed road salt to the recipient, and in higher concentrations than if road salt was transported through the terrain via vegetation, soil and groundwater.

Several studies have shown that a relatively stable salt layer can form in deep stormwater ponds through the winter (Marsalek 2003, Semandeni-Davies 2005, Bækken 2005 and Roseth 2007). Density differences could affect the flow and retention in the pond. Run-off water with low salinity will be able to flow through the dam in the low saline surface layer (Marsalek 2003). The combination of reduced retention and a salt layer may influence sedimentation of particles with traffic caused pollution components, and result in a poorer cleansing effect. Water with a medium salt content will be able to flow through an intermediate layer in the pond, i.e. the layer immediately above the saline and stable bottom water. Saline water will be stored in the salt bottom water with a long retention time.

Pollution containing particles which are stored in salt bottom water are thus expected to have a long retention time and have good time for sedimentation.

In wintertime the mud that has deposited in the stormwater pond is exposed to increased salt concentrations. Several studies have shown that increasing content of salt can lead to mobilization of metals such that these have an increased bioavailability and toxicity (Oberts et al. 2000, Amrhein et al. 1992).

At UMB this issue has been studied in two master's degrees, where one was completed in 2007 (Leistad 2007) and one is under completion. In both of these studies sediments from stormwater ponds along the E6 in Oslo and Akershus have been exposed to water with different salt concentrations in shake-and column experiments. The results showed no clear changes in metal concentrations in the solution phase due to the mobilization of metals from solid phase by exposure to high salt concentrations.

Layout and design of stormwater ponds show considerable variation. In most cases, ponds are established with a sedimentation dam or sedimentation basin in the inlet that is integrated or that has a solid bottom. To ensure the best possible sedimentation, these structures always have an outlet over a solid edge at a high altitude above the bottom of the pool. Similarly, the accumulated run-off water is directed in high up so that the inlet pipe will not be buried by the added sediment. With such a design the adaptation of the sedimentation basin the formation of a salt layer can affect the efficiency of sedimentation both positively and negatively. Positive for particles that are transported in water together with high levels of road salt, and negative for particles that are transported in water with low salinity. The stability of any layer will depend on the size and design of sedimentation basin in relation to the added amount of storm water. Greater run-off episodes with the formation of turbulence will wash with the stable stored salt water.

Furthermore, the stormwater ponds contain a deep main pool or a shallow wetland filter. In ponds with deep main pools different outlet solutions have been designed. Some solutions provide the outlet of the bottom water and some have the outlet of the surface water at the overflow or outlet on the top of the pool. The design will be important both for the degree of cleansing and for the transport of road salt through the stormwater pond. It is uncertain what the best solution is both in terms of cleaning power and release of road salt to a recipient.

Deep and large ponds with run-off in structures close to the surface can lead to a stable salt layer and will be able to store the salt water throughout the winter. In the event of turbulent flow in the entire plant during a spring flood, the salty bottom layer could be washed out in a sudden pulse. Alternatively, it can happen through a slow and gradual washout during the spring, summer and autumn. Which of these events are the most environmentally optimal for the watercourse downstream to the stormwater pond is uncertain and will vary with local recipient conditions.

On behalf of the Norwegian Public Roads Administration, Bioforsk have conducted environmental monitoring of selected stormwater ponds built along the new E6 Oslo border to Svinesund (Norway). In two of the stormwater ponds the in- and outlet quality is clarified through the tapping of water proportional mixed samples. The investigations of a stormwater pond along the E6 at Taraldrud in Ski (Norway) show that there is a salt layer in the pond during the winter (Figure 18). The analysis of chloride in the mixed samples from the pond showed small differences between inlet and outlet, although the highest values were found in the inlet samples (Figure 19). The lowest outlet concentrations of salt are found firstly in late autumn, right before the start of the new salting season (Figures 19 and 22).

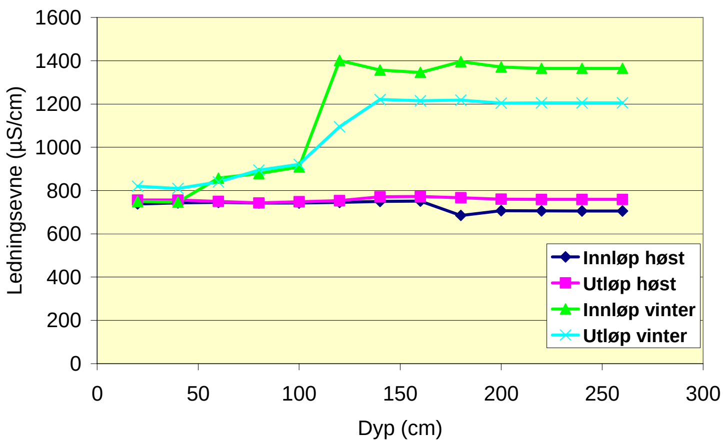

Figure 18*:* **Shows that there is salt layer in the stormwater pond along the E6 at Taraldrud in Ski (Norway) during the winter. Stored heavy saline water is diluted and washed out throughout the summer and in autumn there were no differences in the salinity in the bottom and surface water (Roseth, 2008, unpublished).**

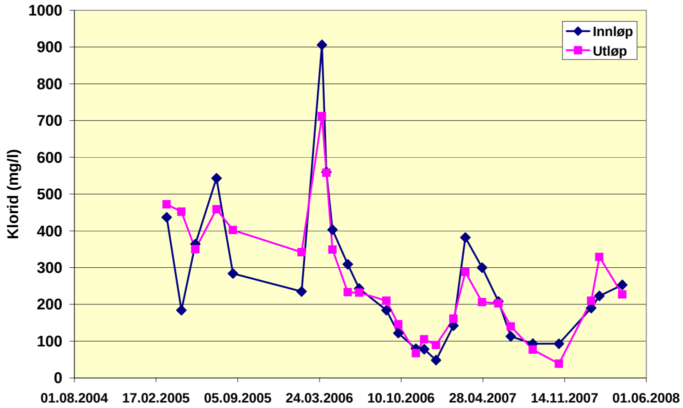

Figure 19*:* **Concentrations of chloride in the inlet and outlet of water from a deep stormwater pond for the treatment of surface water from the E6 at Taraldrud in Ski (Norway) (Roseth, 2008, unpublished).**

A wetland filter is built with a maximum water depth of 0.5 m and is currently considered to be exposed very little to salt layer (Semani-Davies 2005).

Semani-Davies (2005) have studied the function and cleansing effect of a treatment pond for use on run-off water in an urban area. The study showed that the application of road salt had a strong influence on flow conditions in the pool. A markedly salt layer arose after a selective melting of salt from snow, and up to 80 % of the applied chloride was stored in the bottom water of the pool. Stored chloride was washed out again later in the spring when heavy run-off episodes occurred. The study also showed that cleansing efficiency for lead, zinc, and particulates were nearly halved during the winter, while the cleansing effect of copper and cadmium was at about the same level. Other studies have shown a reduced cleansing effect in such pools during the winter. Ice cover, low water temperature and application of road salt are considered as possible explanations for reduced cleansing effect in the winter.

Bækken et al. (2005) have studied the cleansing effect and transport of road salt through a stormwater pond contructed along the new E18 in Vestfold. The investigations showed that water with high levels of road salt lay along the bottom of the pond (Figure 20). The highest salt concentrations were found in the bottom of the pool for sedimentation. Saline bottom water was added water which led to continuously mixing of some of the bottom water. This resulted in equalized salt concentration in the outlet. Beyond spring and summer salt water was stored proportionate to the recipient through the mixing in run-off through the pond (Figure 21). The investigations indicated that the actual retention time for added surface water was lower than the theoretical retention time, and that only part of the water body in the pool was utilized as the active cleansing volume. In future building of stormwater ponds, it has been recommended to consider the establishment of wetland solutions that complement the pool solutions.

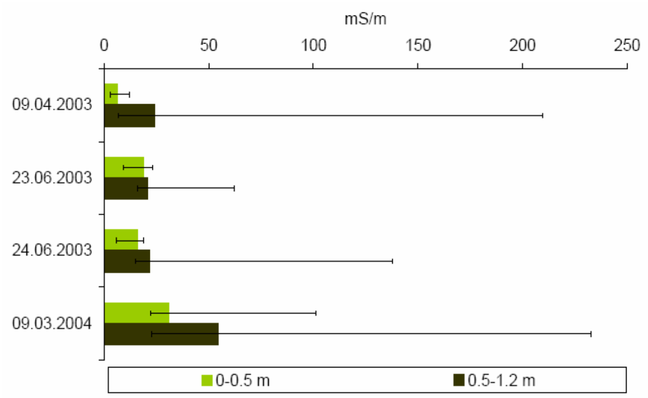

Figure 20*:* **Showing the distribution of salt between the upper (0-0.5 m) and lower (0.5-1.2 m) layer of stormwater ponds along the new E18 in Vestfold (Norway). Indicates average values for the entire basin with a maximum and minimum values at each date (from Bækken et al. 2005).**

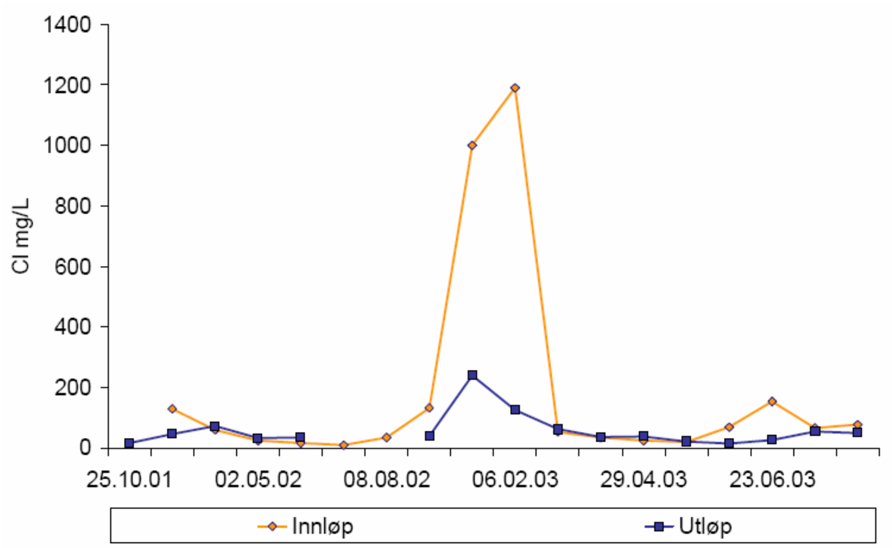

Figure 21*:* **Showing in-and outflow concentrations of chloride in the investigations of a stormwater pond along the new E18 in Vestfold (Norway)(from Bækken et al. 2005).**

Åstebøl (2005) monitored a stormwater pond for run-off water from the E6 Skullerud in Oslo (Norway) with respect to run-off volumes, hydraulic function, purification factor and concentrations of road salt in the inlet and outlet. The study documented a good cleansing rate for all major traffic caused pollutants (60 - 90%), and the purification factor was not significantly reduced through the winter because of ice and salt layer. The study did not focus on road salt, but the conductivity was measured for all samples from the inlet and outlet (Figure 22). Some of the samples were analyzed for chloride. The highest measured chloride concentration from the inlet of the pool was 1830 mg Cl/l and the highest from the outlet was 1810 mg Cl/l (05.02.04). A comparison of results for the measured conductivity and chloride indicates that the maximum chloride concentration added to the pool has been substantially higher, and probably more than 5000 mg Cl/l. The results for conductivity from the samples from the treatment pool demonstrates very high concentrations of road salt in the inlet for some of the samples related to a combination of salting and snow melt. Based on conductivity the concentrations of applied road salt seem to be equalized in the outlet water from the stormwater pond. The results also indicate that the salt accumulated in the pool through the winter season are released gradually through the summer and contribute to the elevated salt concentrations up to the end of July. The lowest conductivity in the outlet was found in October, just before the start of the new de-icing season.

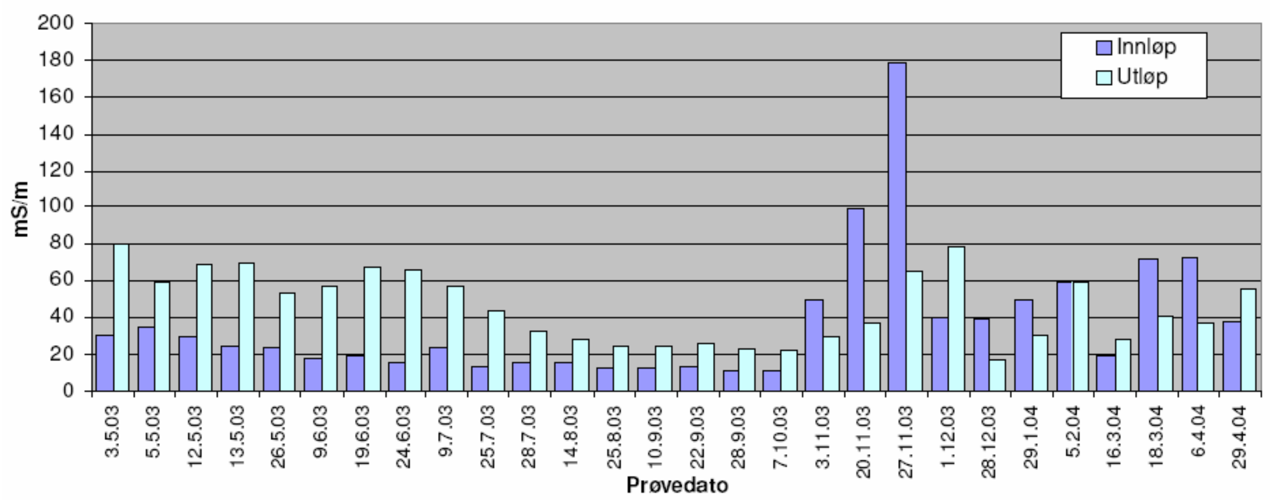

Figure 22: Shows the conductivity of samples from the input and output of a stormwater pond for surface water from E6 at Skullerud junction in Oslo in 2003 and 2004 (From Åstebøl 2005).

Many publications describe run-off patterns and particle transport associated with snow melt (Marsalek et al. 2003). In the snow a selective melting out of ionic, polar and water-soluble compounds occurs. The majority of road salt and soluble pollution components in the snow will therefore be leached through the snow and leak out with the first melt water. The first melt water (first flush) has a high content of salt and water soluble pollution compounds, but also has a high density. In applied basins and stormwater ponds such salt melt water will contribute to the formation of a stable salt layer.

Un-polar compounds as PAHs and oil and contaminated particles will be stored in the snow until the end of snow melting and will be initially mobilized with the last melt water. A large proportion of the traffic created pollution components stored in the snow will therefore be transported out of poor ionic run-off water at the end of snow melting. In a climate where snow melt is a dominant part of the run-off in the spring, the combination of the salt layer in stormwater pond and large pollution transport in water with low conductivity could pose a challenge in relation to the effective cleansing of pollutions.

12.6 Alternative de-icing chemicals and stormwater ponds

In Norway, there has been conducted few tests of alternative de-icing agents on the road. In cooperation with relevant contractors have the Norwegian Public Roads Administration conducted practical experiments with road salt added to Magnesium chloride (www.vegvesen.no) or the Magnesium chloride in combination with an organic "Syrup-like" compound (Vegen og vi 17/06). Mesta performed experiments with "syrup" (corn starch) in combination with Magnesium chloride on the E6 north of Oslo season 06/07.

In a test project conducted over 5 years, has magnesium chloride been added at 3 percent of the weight of the regular road salt (NaCl) used in Ring 3 in Oslo. On the basis of good experiences routine the use of road salt with added magnesium chloride is planned on some stretches of roads. Such a practice is expected to result in fewer changes related to the function and degree of purification of stormwater pond compared with the use of regular road salt. Increased application of magnesium will eventually cause minor changes in the sedimentation characteristics of particles, in the form of increased flocculation and increased removal/sedimentation of particles.

Magnesium is a more effective ion exchange component than sodium, and an increased quantity of magnesium in run-off from the road will theoretically be able to mobilize traffic caused heavy metals from particles along the road to the run-off water (see chapter on de-icing chemicals and soil). Completion of the project has not included the collection of environmental information related to any changes in run-off water quality. On the E6 northwards from Oslo tests have been carried out of magnesium chloride in combination with "syrup" from corn production. Preliminary results indicate that this combination can reduce the total salt consumption. Studies on the changes in the composition of run-off water have not been undertaken. Run-off water is expected to contain increased amounts of easily degradable organic matter, which could provide anaerobic conditions in stormwater pond, and especially when the salt layer that is maintained for a period of time during the early summer. This could affect mobilization of traffic caused pollution substances in the mud deposited in stormwater pond.

Reviews of environmental consequences of the use of different types of de-icing agents, including urea, acetate - and formate based agents related to the use of the road de-icing agents used at Norwegian airports have been completed. In an overall assessment for Norwegian airports have they chosen to use formate based road de-icing agents, and this is mainly motivated by the fact that formate has the lowest oxygen consumption during decomposition in soil and water while the use purposes, have properties that are comparable with acetate. Previously, was urea the dominant road de-icing agent, but is no longer due to high oxygen demand associated with decomposition and nitrification and has poorer corrosion properties than the agents based on formate and acetate.

Internationally, CMA (calcium magnesium acetate) is the most widely used alternative to road salt and it a series of studies on the effects on both the friction and the environment have been conducted.

In Sweden there have also been tests on the application of organic matter (waste product from sugar production) to increase the effective time of the regular road salt and make it more durable against leaching (Gustafsson and Gabrielsson 2006 and Thunqvist 2007).

12.6.1 Assessment of oxygen consumption

In the following section a simple assessment of oxygen consumption and the risk of oxygen-free conditions in stormwater pond and recipients on the basis of use of formate and acetate based de-icing agents has been made. The assessment is based on experience with the use of these chemicals for road de-icing at airports.

Normal application when de-icing a runway is 5000 liters Aviform L_{50} (50% Potassium formate) for 2.5 km runway. The solution contains 26 % Formate. Thus there is used about 500 kg pure Formate per km. A runway is about 50 m wide, while a 4-lane highway has a total asphalt width of about 20 m. When converted one will be expected to use around 200 kg of Formate per km. Diluted in 10 mm of rainfall the run-off from the road can theoretically contain 1000 mg Formate per liter, which is equivalent to a theoretical oxygen consumption of 340 mg/l and a biological oxygen demand of 270 mg/l. This calculation assumes that Formate is used in similar quantities as to keep a “bare runway" at an airport. The use of formate based deicing agents used on the roads may be reduced.

If formate is used at concentrations equivalent to the current lower concentrations of chloride, can one expect concentrations of 100-200 mg formate in the road run-off. Oxygen consumption related to formate in run-off water will at these concentrations be around 30 mg/l or higher, and exceed the normal oxygen content available for decomposition in stagnant water (12-15 mg oxygen/l). Formate will therefore create oxygen-free conditions in treatments pools, and especially in a bottom layer with stable stored saline. The risk of oxygen-free conditions will be greatest in the spring, with higher water temperatures that result in a faster decomposition of the applied formate.

Acetate has a higher oxygen consumption than formate (1.07 to 0.34 mg oxygen/mg component), and will constitute a bigger threat in relation to oxygen consumption and anaerobic conditions provided that they are applied to the road in comparable amounts. Experiences from their use at airports indicate that consumption of formate and acetate are comparable.

12.7 Comprehensive discussion – run-off water management and road salt

The review of run-off water management and road salt has focused on the following issues:

- The current practice for run-off water management, and the effects in recipients vulnerable to the application of salt
- Design of a stormwater ponds and effects on storage and leaching of salt
- Salt layering in a stormwater pond and the effect on retention time and cleansing processes
- Protection measures related to the dispersal and dilution or directing of saline run-off water
- To what degree do the environmental impact assessments for new high-traffic roads take into consideration road salt and the opportunities to prevent salt created damage to recipients
- Opportunities to remove and equalize salt pulses in urban environments
- Stormwater ponds, recipients and the use of the de-icing agents with organically biodegradable components.

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