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# Evolution of tunnel wash water quality during sedimentation

Master's thesis in Civil and Environmental Engineering
Supervisor: Thomas Meyn
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NTNU
Norwegian University of Science and Technology

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# Preface

This master's thesis was submitted as the final contribution in the Civil and Environmental Engineering program at the Norwegian University of Science and Technology (NTNU), in the subject *TVM4905 - Water Supply and Wastewater Systems, Master's Thesis*. This thesis continues the work of academic supervisor Thomas Meyn, former master students at the Department of Civil and Environmental Engineering and the consulting firm ViaNova Trafikk AS, for the Norwegian Public Roads Administration (NPRA). The aim is to expand the knowledge of the sedimentation process as a treatment step for tunnel wash water. Determination of pollutant groups Polycyclic Aromatic Hydrocarbons (PAHs) and detergents in samplings from this thesis are studied in the on-going PhD thesis of H. Vistnes at the Department of Civil and Environmental Engineering, NTNU (expected published by 2024). Therefore, the reader may find that these parameters are lacking from the Results section. Parts of the literature review for the Background chapter (2.1 and 2.4) was performed autumn 2020, as part of the subject *TVM4510 - Water and Wastewater Engineering, Specialization Project*.

Special thanks to my academic supervisor Associate Professor Thomas Meyn and PhD candidate Hanne Vistnes for all academic and practical guidance. Also, thanks to:

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- Mesta's and Norwegian Public Roads Administration's departments in Trondheim, for allowing visitors during their tunnel washing, for allowing sampling in and for answering my questions about the practical sides of tunnel washing.

Trondheim, June 17, 2021

Sofie Eivik Karlsen

# Abstract

Tunnels are regularly washed to ensure lifelong performance of technical equipment. Traffic safety is also promoted through sufficient visibility inside the tunnels. Tunnel washing produces large volumes of contaminated tunnel wash water regularly, containing elevated concentrations of several pollutants compared to road runoff; heavy metals, polycyclic aromatic hydrocarbons (PAHs), detergents, and organic tire components. Hence, the tunnel wash water discharge needs to be handled so that it does not cause disproportionate harm to the aquatic environment in the recipient.

Sedimentation is commonly implemented as a primary treatment step in highly trafficked tunnels in Norway (Average Annual Daily Traffic $>$ 15 000) to reduce the environmental harm of tunnel wash water discharge. This Master's thesis aims to increase the available knowledge on how tunnel wash water quality evolves during the sedimentation process and what tunnel wash water quality sedimentation yields. Tunnel wash water was therefore sampled from tunnels on urban highways in Oslo and Trondheim. Selected physicochemical characteristics of sampled tunnel wash water were measured during 30 days of sedimentation in a custom-made sedimentation pilot of 2 m height.

The results show a significant decrease in particle load and particulate heavy metal concentrations. Turbidity, total suspended solids ($>$ 0.45 $\mu$m), and the total number of particles declined ranged from 94 to 99 % during sedimentation. The removal efficiency of the particulate fraction of heavy metals ranged from 49 % for zinc to 99.7 % for aluminum. Principal component analysis demonstrated that particulate heavy metal and TSS removal were strongly correlated; p-values of 0.91 to 1.0 for seven out of ten studied metals (Al, As, Cu, Cr, Fe, Mn, and Ni).

Despite relatively high removal efficiency, Cu, Zn, As, and Cr effluent concentrations in the studied tunnel wash water exceed recipient threshold concentrations for short time exposure. This implies that discharge of tunnel wash water from the studied tunnels can cause environmental harm. Furthermore, the bulk of effluent Cu, Zn, and As concentrations are in the bio-available, dissolved form. The dissolved fractions of metals, organics, ions, and particles smaller than 0.1 to 1.0 $\mu$m hardly reduced during sedimentation. Therefore, the design of potential secondary treatment steps should remove the persistent dissolved matter. Furthermore, the recipient dilution effects and vulnerability need to be classified to design primary and secondary treatment steps with appropriate treatment efficiency. Lastly, asset management of the studied primary treatment facilities was found to be of varying quality. The prerequisite to achieve the desired treatment efficiency, i.e., that the assets can deliver their function, hence lack in individual facilities in both Oslo and Trondheim.

**Keywords:** Heavy metal removal, heavy metal fractionation, particle size distribution, principal component analysis

# Sammendrag

Tunneler vaskes regelmessig for å sikre at tekniske installasjoner kan fungere optimalt og vedlikeholdes gjennom brukstiden. Fjerning av svevestøv og belegg på tunnelens overflater og belysning fremmer også trafikksikkerhet, ved å sørge for tilstrekkelig visuell ledning for trafikkanter. Tunnelvasking produserer store volum med forurenset tunnelvaskevann regelmessig, som inneholder høye konsentrasjoner av forurensende stoffer, sammenlignet med vegavrenning; tungmetaller, polysykliske aromatiske hydrokarboner (PAHer), rengjøringsmidler og organiske forbindelser fra bildekk. Derfor må utslipp av tunnelvaskvann håndteres slik at det ikke forårsaker uforholdsmessig skade på vannmiljøet i resipienten.

Sedimentasjon blir ofte brukt som primærrensetrinn i høyt trafikkerte tunneler i Norge (årsdøgntrafikk $>$ 15 000) for å redusere miljøskadene ved utslipp av tunnelvaskevann. Denne masteroppgaven tar sikte på å øke den tilgjengelige kunnskapen om hvordan tunnelvaskvannskvaliteten utvikler seg gjennom sedimenteringsprosessen og hvilken vannkvalitet sedimenteringen gir for tunnelvaskevann. Prøver av tunnelvaskevann ble derfor tatt fra tunneler på urbane motorveier i Oslo og Trondheim. Utvalgte fysisk-kjemiske parametere ble målt for prøvene i løpet av 30 dagers sedimentering i en 2 m høy, stedsbygd sedimenteringspilot.

Resultatene viser en betydelig reduksjon i partikkelmengde og partikulært bundet tungmetaller. Turbiditet, total suspenderte stoff (TSS; $>$ 0.45 $mu$m), og det totale antallet partikler ble redusert i spennet fra 94 til 99 % i løpet av sedimentering. Renseeffektiviteten for partikulære fraksjoner av tungmetaller varierte fra 49 % for sink til 99,7 % for aluminium. Statistisk analyse viste at fjerning av partikulært bundet tungmetall og TSS var sterkt korrelert; p-verdier på 0,91 til 1,0 for syv av ti studerte metaller (Al, As, Cu, Cr, Fe, Mn og Ni).

Til tross for relativt høy renseeffektivitet, overskrider Cu, Zn, As og Cr i prøvetatte tunneler veiledende resipientkonsentrasjoner for korttids-eksponering. Dette betyr at utslipp av tunnelvaskvann fra de studerte tunnelene kan forårsake miljøskade i resipientene. I tillegg er hovedparten av Cu, Zn, og As konsentrasjonene i oppløst form, som er mer biotilgjengelig enn partikulær form. De oppløste metallene, organiske stoffene, ioner og partikler som er mindre enn 0.1 til 1.0 $\mu$m reduseres lite i løpet av 30 dagers sedimentering. Derfor bør utformingen av potensielle sekundære rensetrinn fjerne det bestandige, oppløste stoffet. Videre bør resipientenes fortynningsevne og sårbarhet klassifiseres for å utforme primære og sekundære rensetrinn med passende renseeffekt. Som en sluttbemerkning ble forvaltningen av de studerte primærbehandlingsanleggene funnet å ha varierende kvalitet. Forutsetningen for å oppnå ønsket behandlingseffekt, dvs. at anleggene kan levere sin funksjon, er derav manglende for fasiliteter i både Trondheim og Oslo.

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# Terminology

| | |
|---|---|
| **AA-QS** | Annual Average concentration - Quality Standard |
| **AADT** | Average Annual Daily Traffic |
| **DOC** | Dissolved Organic Carbon |
| **EC** | Electric Conductivity |
| **EQS** | Environmental Quality Standards |
| **GfT** | Granfoss tunnel |
| **GT** | Grillstad tunnel |
| **IPC-MS** | Inductively coupled plasma mass spectrometry |
| **kDa** | Kilo Dalton |
| **LOQ** | Limit of Quantification |
| **MAC-QS** | Maximum acceptable concentration - Quality Standard |
| **N/A** | Not Available |
| **NPRA** | Norwegian Public Roads Administration (Nor.: Statens Vegvesen) |
| **PAH** | Polycyclic Aromatic Hydrocarbons |
| **PIDS** | Polarization Intensity Differential Scattering |
| **PSD** | Particle Size Distribution |
| **ROI** | Residue on ignition |
| **SLES** | Sodium lauryl ether sulfate |
| **SmT** | Smestad tunnel |
| **Std. Dev.** | Standard Deviation |
| **ST** | Strindheim tunnel |
| **TSS** | Total Suspended Solids |
| **TS** | Total Solids |
| **TT** | Tåsen tunnel |
| **TWW** | Tunnel Wash Water |
| **US EPA** | The United States Environmental Protection Agency |
| **WFD** | Water Framework Directive |

# 1 Introduction

Norway has more than 1100 road tunnels, making them a characteristic element for Norwegian roads. Tunnels allow reliable transport to Norway's many dispersed settlements. They also shield the urban environments from traffic noise and pollution if properly maintained (Meland et al., 2010). Highly trafficked tunnels (Average Annual Daily Traffic $>$ 15 000) are washed up to once a month to remove dust, road salt, debris, and oily coatings that accumulate inside tunnels. Routinely washing produces large volumes of tunnel wash water, containing pollutants originating from vehicle and road surface abrasion, combined with vehicle combustion. The same pollutant sources appear in road runoff, but the particle accumulation inside tunnels between washing events differentiate the two (Meland and Rødland, 2018). Pollutants are detained by the tunnel cross-section, including air-borne pollutants, while road runoff is subject to meteorological variations in wind and precipitation. In consequence, tunnel wash water generally contains elevated concentrations of traffic-generated pollutants, including heavy metals, polycyclic aromatic hydrocarbons (PAHs), and organic tire components (Meland and Rødland, 2018; Hallberg et al., 2014; Klöckner et al., 2021). Although, the pollutant concentrations vary significantly in both road runoff and tunnel wash water. Lastly, the detergents used to wash tunnels distinguish tunnel wash water from road runoff.

The sum of various pollutants present in tunnel wash water raises concerns for the recipients' chemical and biological quality. Poor aquatic environments can originate from heavy metals and PAHs concentrations, combined with toxic detergent components, and oxygen deficiency from bio-degradation of organic compounds (Johansen and Thygesen, 2013; Meland, 2010; Ying, 2006). Nevertheless, most of Norway's tunnels have no treatment solution before regularly discharging large volumes of tunnel wash water in order of magnitude of 100 $m^3$ per km of washed tunnel (Torp and Meland, 2013). Consequently, the potential environmental harm is pressing in the water bodies frequently receiving tunnel wash water discharge.

In recent years, treatment facilities for tunnel wash water have been constructed to lower the risk of environmental harm in recipients (Meland, 2012). The current treatment guidelines for road runoff and tunnel wash water include recipient vulnerability as a design criterion. Meaning the most vulnerable recipients require more treatment of tunnel wash water before discharge. In addition, traffic amount is included due to the correlation between higher traffic amounts and higher vehicle and road abrasion. These criteria aim to promote recipient water quality. However, the treatment requirements lack thresholds for measuring sufficient treatment efficiency. That is, what effluent water quality primary and secondary treatment facilities should deliver.

Sedimentation basins have been a favored treatment solution for heavily trafficked tunnels. The frequent implementation is due to the potential for reducing particulate bound pollutants and sufficient hydraulic residence time allow degradation of organic compounds (Hallberg et al., 2014). Meaning that reduction of the particulate bound pollutants and detergent degradation is expected during the residence time, but the extent is not classified. Furthermore, the actual treatment efficiency delivered in the built treatment facilities are not documented for tunnel wash water. Åstebøl et al., 2013, found that 21 out of 26 studied wet sedimentation ponds receiving road runoff could not deliver satisfactory treatment efficiency. This was due to deviations from design in executed construction and insufficient operation and maintenance. These findings suggest that the situation for tunnel wash water sedimentation basins may be critical as well.

The previous studies and reports addressing tunnel wash water quality mainly study its untreated quality (Meland et al., 2010; Meland, 2012; Meland and Rødland, 2018; Johansen and Thygesen, 2013; Allan et al., 2016; among others). Fewer studies have documented tunnel wash water treatment efficiency during the most common treatment step; sedimentation. Furthermore, published studies on sedimentation of tunnel wash water have mainly been on small-scale volumes (Aasum, 2014; Garshol, 2016; Roseth and Søvik, 2006; Hallberg et al., 2014). Previous master theses Bjotveit, 2020, and Kowollik, 2020, performed larger scaled sedimentation experiments (140 to 280 L). These theses initiated the documentation of tunnel wash water quality changes during sedimentation, for the on-going research project of the Norwegian Public Roads Administration (Nor.: Statens Vegvesen). Still, the current data basis is too scarce to properly design primary and secondary treatment steps for tunnel wash water. In addition, the processes occurring in the tunnel wash water during sedimentation is not fully understood. Therefore, the same research project continues in the current thesis to better understand sedimentation as a primary treatment step for tunnel wash water. Expanded knowledge of sedimented tunnel wash water quality is then a prerequisite for designing a secondary treatment step. Finally, the research project as a whole aims to equip the authorities with a future overall treatment strategy. In this context, the present thesis aims to answer the following research questions:

- How does 30 days of sedimentation affect tunnel wash water quality, considering parameters for particulate bound and dissolved contaminants?
- What does the final, sedimented tunnel wash water quality imply for authorities when designing primary and secondary treatment steps?

## 1.1 Structure of this thesis

Following this introduction is the Background chapter describing tunnel washing. In addition, the same chapter describes detergent components and heavy metals frequently appearing in tunnel wash water. Followed by a review of the current treatment requirements for tunnel wash water. The subsequent method chapter explains why and how water quality parameters were measured and the statistical analysis of the measurements. Successively, the presentations of results and discussion are combined in one chapter. Finally, the conclusion and recommendations for further work concludes this thesis.

# 2 Background

The current Background chapter intends to equip the reader with information that may not be evident without reproducing previously written theses. More fundamental, detailed background theory chapters have previously been presented in theses in the same project as the current thesis (Bjordal, 2020; Kowollik, 2020; Bjotveit, 2020), in addition to the former thesis of Aasum, 2014.

## 2.1 Washing procedure of tunnels in Norway

The Norwegian Public Roads Administration (NPRA) gives minimum requirements for washing frequencies of Norwegian tunnels proportional to increasing average annual daily traffic (AADT) (*Håndbok R610 Standard for drift og vedlikehold av riksveger,* 2014, p. 66). The requirements are given to achieve the objectives of washing tunnels, i.e., safe conditions for road users and a life-prolonging environment for technical equipment. Tunnels with AADT higher than 15 001 vehicles per day are classified uniformly by the NPRA as the most heavily trafficked group. Tunnels with AADT higher than 15 001 vehicles per day have minimum required washing frequency as follows: 2 *whole* washes, 4 *half* washes and 6 *technical* washes pr. year.

Minimum required washes are distributed over the year according to local variations and agreements between contractors and the NPRA. For example, periods with frost through tunnels hinder washing, causing postponed washes or dry brushing as a replacement. During the same periods, which is also studded tires season, densely trafficked tunnels (AADT $>$ 15 001 vehicles/day) are often additionally dry swept in-between washes to control dust amounts. Extended frost periods are accounted for in the yearly spring whole washes of tunnels, where water and brushing is utilized. In summary, inter-event pollutant accumulation varies with traffic and local climate, from less than one month for high-trafficked tunnels (AADT $>$ 15 001) and up to one year for the least trafficked tunnels (AADT $<$ 300 vehicles/day).

Figure 1: Illustrative tunnel road surface before and after wash, showing improved visibility of road markings. Photos: Norwegian Public Roads Administration, Trondheim 2016 (Midtbø, 2016).

*Whole wash* (Nor: helvask) involves washing all parts of the tunnel's cross-section: roadway, hard shoulder, roof (crown), walls and all technical installations (lighting, cable bridges, doors, emergency equipment and similar). *Half wash* (Nor: halvvask) is the same procedure, excluded the tunnel's roof, or above 3.5 m on the walls if the transition from wall to ceiling is unclear. *Technical wash* (Nor: teknisk vask) includes wash of roadway, hard shoulder and all technical installations inside the tunnel (*Håndbok R610 Standard for drift og vedlikehold av riksveger,* 2014).

In advance of *whole* and *half* washes, the roadway and hard shoulder are swept, i.e., reducing the amount of coarse pollutant load in the tunnel wash water being drained to the recipient or treatment solution. After sweeping the tunnel floor, walls are sprayed with detergent. The detergent is then allowed to react with the accumulated solids before being sprayed with water (Figure 2) and brushed away by specially equipped tankers. Tunnel washes ends with excess water and remaining matter being swept from the tunnel's road surface by specially designed suction vehicles (Meland, 2012). Consequently, not all tunnel wash water reaches the treatment or recipient, e.g., in the Smestad tunnel (Oslo), 80% of the tunnel wash water is estimated to reach the sedimentation pond (Frost and Gremmertsen, 2019). Similarly, a single study of the Nordbytunnelen (Oslo) measured 77% runoff from a half wash of the tunnel (Garshol, 2016). Both estimates comply with previous findings of 70 to 90% collection in Norwegian tunnels' drainage systems (Meland, 2012).

Figure 2: Tanker with nozzle equipment, washing shotcrete tunnel lining of the Grillstad tunnel in Trondheim, Norway (Photo by T. Meyn, 26.03.21).

Local variations in tunnel constructions affect the way they are washed. Typically, shotcrete and blasted rock tunnel linings have the potential to accumulate pollutants in the uneven surface structure. Consequently, such tunnel linings are washed utilizing low-pressure nozzles ($<$ 15 bars), aiming at optimal cleaning effect. Opposite, smoother surfaces of prefabricated concrete would more often be washed using high-pressure nozzles ($>$ 100 bars). During the same washing event, both types of nozzles may be utilized, to achieve optimal cleaning efficiency of individual part of the tunnels' cross section (Mesta Trondheim, personal communication 25.03.21). Differences in nozzle pressure also lead to differences in water consumption between tunnels. Previous studies indicate that high pressure nozzles consume less water per meter tunnel cross section washed, compared to low pressure nozzles (Torp and Meland, 2013).

## 2.2 Detergents in tunnel wash water

Detergents are often used when washing tunnels, to enhance washing efficiency. Even for low concentrations ($< 1\%$), detergents enhance the cleaning efficiency, compared to solely using water (Ying, 2006). The composition of each detergent type varies with producer, but some commonly used active ingredients can be listed: *surfactants (surface active agents)*, *complexing agents*, *alkaline substances* (or *Silicates*) and *solvents* (Roseth and Søvik, 2006).

*Surfactants* have traditionally been the main detergent component of concern for acute and chronic toxic properties in elevated recipient concentrations. Therefore, surfactants are described in more depth than the rest of detergents' active ingredients. Surfactants are defined as "*surface active compounds that have a propensity to sorb to sediments and suspended particles that eventually settle and form sediments*" (Dyer et al., 2006, p.46). Surfactants generally consist of a polar head group, which is hydrophilic (well solvable in water), and a non-polar, hydrophobic hydrocarbon tail (not solvable in water). Combined hydrophobic and hydrophilic molecular parts allow surfactants to break the water surface tension by micelle formations. Micelles constitute of a centre of the hydrophobic hydrocarbon tails, where non-polar pollutants can be detained. The hydrophilic end is then in contact with the surrounding water. This enhance detergents' penetration of pollutants.

The charge of the surfactant's hydrophilic end determines the classification of the surfactant: cationic, non-ionic or anionic. The respective order also represents the surfactants' sorption affinity for particles, sludge and sediments (Ying, 2006). In addition, surfactants' head groups can carry both positive and negative charge, called zwitterionic or amphoteric surfactants (Rapp, 2017). Mixtures of more than one surfactant type are commonly contained in detergents to optimize for efficient removal of different kinds of accumulated dust and coatings inside tunnels.

In addition to having sorption-induced settling potential, surfactants can be biodegraded. Surfactant degradation is characterized as a stepwise process, with varying documentation of intermittent degradation steps for different detergents. Succeeding intermittent degradation, ultimate degradation occurs aerobically when surfactant molecules are mineralized to $CO_2$, water, mineral salts and biomass (Barra Caracciolo et al., 2017). Surfactant degradation lowers the environmental risk of its components. If cationic and anionic surfactants are released untreated, they induce chronic toxicity at recipient concentrations greater than 0.1 mg/L (Ying, 2006). The environmental risk of surfactants is determined based on discharged surfactant concentrations and its bio-availability. Those two variables again determine if discharge concentration can obtain equilibrium with the recipient's sludge or biofilm natural removal rate of surfactants (Dyer et al., 2006).

*Complexing agents* complex with water hardness ions ($Ca^{2+}$, $Mg^{2+}$) and metal ions, to hamper metal interference with surfactant micelles (Roseth and Søvik, 2006). *Alkaline substances* have the primary purpose of sustaining basic pH in the wash water to facilitate emulsifying of oily compounds, predisposing them to micelle formation. *Solvents* are mainly organic, with the ability to remain chemically stable while dissolving non-polar components, e.g. oily components in tunnels. Meaning solvents contribute with an added emulsifying process, inhibiting grease and oil from recombining after being emulsified by alkaline substances (Firestone and Gospe, 2009).

### 2.2.1 Purify Infravask detergent

Tunnels in the Oslo area are washed using Purify's detergent *PG-IV1-X1-1000*, also called *Infravask*. This detergent's main constituents of interest are 5 - 15% anionic surfactants, with Benzenesulfonic acid (LAS; $CH_3[CH_2]_nCH_2CHSO_3Na^+$, $n = 10 - 13$) composing $<$ 5% of those, and $<$ 3% sodium lauryl ether sulfate (SLES; $CH_3[CH_2]_{11}[OCH_2CH_2]_nOSO_3Na^+$, $n$ not further specified (Barra Caracciolo et al., 2017)). Both surfactants are aquatic toxic in undiluted form, but the toxicity decreases proportionally to dilution degree in water. Full safety data sheet, as given by producer, can be found in Appendix C. The surfactant properties of the Infravask detergent are summarized in Tab. 1.

### 2.2.2 Sanego Solvynol Truck detergent

Tunnels in the Trondheim area are washed using Sanego Solvynol Truck detergent. It contains 1 to 5% of both the non-ionic surfactants ethoxylated fatty alcohol ($C_{10}H_{21}[OCH_2CH_2]_nOH$, $n = 0 - 8$) and alkoxylated fatty alcohol ($[C_3H_6O.C_2H_4O]_nC_{10}H_{22}$, $n$ not further specified). In addition, it contains 0 to 1% of the zwitterionic surfactant Cocoamidopropyl betaine ($CH_3[CH_2]_{10}CONH[CH_2]_3N^+[CH_3]_2CH_2CO_2^-$), having both a Quaternary ammonium *cation* and a carboxylate (anionic). The safety data sheet (App. D) also states 1 to 5% content of the alkaline substance sodium hydroxide (NaOH), increasing the pH of the tunnel wash water.

Table 1: Properties of detergents Purify Infravask, used for washing tunnels in the Oslo area, Norway, and Sanego Solvynol, used for washing tunnels in the Trondheim area, Norway. Specifications are given by the respective producers Purify (App. C) and Sanego (App. D). *In pure form, harm potential decreases as components are diluted with water.

| Surfactant component | | CAS-nr. | Content (%) | Environmental classification* |
|---|---|---|---|---|
| **Purify Infravask** | Benzenesulfonic acid (LAS) | 68411-30-3 | $< 5\%$ | Acute toxic (oral ingestion), Aquatic chronic toxic |
| | Sodium lauryl ether sulfate (SLES) | 9004-82-4 | $< 3\%$ | Aquatic chronic toxic |
| | Total surfactant content | | 5 to 15% | |
| **Sanego Solvynol** | Ethoxylated fatty alcohol | 160875-66-1 | 1 to 5% | Acute toxic (oral ingestion) |
| | Alkoxylated fatty alcohol | 166736-08-9 | 1 to 5% | Acute toxic (oral ingestion) |
| | Cocoamidopropyl betaine | 147170-44-3 | $\leq 1\%$ | - |
| | Total surfactant content | | N/A | |

## 2.3 Heavy metals in tunnel wash water

Heavy metals discharge poses a threat for the development of toxic recipient environments due to their "*bio-accumulative potential, non-biodegradable properties and toxicity of the contaminants even at low concentrations*" (Mouni et al., 2009). Heavy metals may enter an organism's body through consumption, drinking water, or inhaling, initiating bio-accumulation that may be carcinogenic and harm organisms' nervous system (Bonilla-Petriciolet and Mendoza-Castillo, 2017). Low molecular mass metal species have been indicated to have the highest mobility and bio-availability (Meland et al., 2010). In contrast to particulate heavy metal fractions, low molecular mass fractions are also less likely to be removed by gravity-induced sedimentation. Primary treatment by sedimentation is still justified by the dominating particulate fraction of heavy metals in road runoff and tunnel wash water, typically constituting 40 to 90% of the total heavy metal concentrations (Meland, 2012).

Table 2: Heavy metals frequently appearing in tunnel wash water, as found in the literature review by Meland, 2010. Supplemented by *(Adamiec et al., 2016).

| **Metal** | **Source** |
|---|---|
| Aluminium (Al) | Brakes*, tires, road surface |
| Arsenic (As) | Asphalt wear* |
| Cadmium (Cd) | Tires, combustion |
| Chromium (Cr) | Brakes*, vehicle body, combustion |
| Copper (Cu) | Brakes, tires |
| Iron (Fe) | Brakes, vehicle body, road surface |
| Lead (Pb) | Brakes, tires, road surface |
| Magnesium (Mg) | Road surface |
| Manganese (Mn) | Tires |
| Nickel (Ni) | Brakes, combustion, asphalt |
| Zinc (Zn) | Combustion, bitumen*, tires*, road equipment |

Heavy metals previously analysed in tunnel wash water coincide with the United States Environmental Protection Agency (US EPA) defined pollutants of interest; As, Cd, Cr, Cu, Pb, Ni, Zn (Miljødirektoratet, 2018). In addition, Al, Fe, Mg and Mn are occasionally reported (Meland et al., 2010; Meland and Rødland, 2018; Garshol, 2016; Aasum, 2014; Andersen and Vethe, 1994). These heavy metals have been found to originate from different parts of the vehicles, their combustion and their interaction with the road surface. A summary of the above-mentioned heavy metals' origin is given in Tab. 2, while complete tables have been provided in previous master theses Aasum, 2014, Kowollik, 2020, Bjordal, 2020 from the literature review of Meland, 2010. The table emphasizes how the vehicle and road surface have been found to be main contributors to heavy metal occurrence in road dust. Furthermore, vehicle speed and repeated breaking and acceleration have been indicated to cause higher heavy metal concentrations in road dust (Adamiec et al., 2016).

## 2.4 Implementation of current treatment requirements

In 2007, the EU's Water Framework Directive's (WFD) was implemented in Norwegian law (Vannforskriften, 2007). The WFD's overall objective is to ensure good chemical and biological water quality in European water bodies. The thresholds diverting *good* water quality (no toxic effects) from *moderate* water quality (chronic effects of long term exposure to a substance) are given as the European Quality Standards (EQS) for compounds of interest in water. The EQS are divided into two principal values; annual average concentration quality standard (AA-QS) and maximum acceptable concentration quality standard (MAC-QS). The EQS thresholds are separated to provide protection for chronic and acute exposure, respectively (Miljødirektoratet, 2018). The EQS thresholds are further differentiated between dissolved concentration thresholds ($< 0.45$ $\mu$m) for Cd, Pb and Ni, which is also defined as the bioavailable fraction. Diversely, Cr, Cu, Zn and As thresholds are given for the total metal concentrations (Vannforskriften, 2007). Several of the compounds of interest frequently appear in tunnel wash water, among them polycyclic aromatic hydrocarbons (PAHs) and heavy metals Pb, Cd, Ni, As, Cu, Cr and Zn (Miljødirektoratet, 2018).

The term polycyclic aromatic hydrocarbons (PAHs) consists of numerous compounds, with a common feature of minimum two fused aromatic benzene rings (Meland et al., 2010). In tunnel wash water, PAHs mainly originate from vehicles' tires, combustion, oil or petroleum spill and bitumen in road surfaces. Generally, they can pose a threat to aquatic organisms both as short, acute toxic chains of hydrocarbons and as long, carcinogenic chains (Grung et al., 2016; Meland et al., 2010). The The United States Environmental Protection Agency (US EPA) defined 16 PAHs as compounds of interest, as a response to their potential for environmental harm. Those 16 PAHs have been implemented in the Norwegian regulations as well (Vannforskriften, 2007).

The Norwegian Public Roads Administration (NPRA) has set guidelines for treatment of the pollutant sources road runoff and tunnel wash water. The NPRA thereby aims to "*fulfill its obligations under the Water Regulations, White Paper no 14 2006–2007 and the sector responsibility*" (*About NORWAT*, 2016). Therefore, treatment requirements are set based on traffic density (AADT) and recipient vulnerability (Vegdirektoratet, 2018), as summarized in Tab. 3 below.

Table 3: Summary of treatment requirements for *road runoff*, also used for tunnel wash water, given by Vegdirektoratet, 2018. *Step 1: One treatment step required, removing particulate matter. **Step 1 + 2: Two treatment steps are required, removing particulate and dissolved matter.

|  | **Vulnerability of recipient** |  |  |
|---|---|---|---|
| AADT (vehicles/day) | **Low** | **Moderate** | **High** |
| < 3000 | None | None | None |
| 3000 - 15 000 | None | Step 1* | Step 1* |
| 15 000 - 30 000 | None | Step 1 + 2** | Step 1 + 2** |
| > 30 000 | Step 1 + 2** | Step 1 + 2** | Step 1 + 2** |

Sedimentation is the most commonly utilized primary treatment step for tunnel wash water in Norway ("Step 1" in Tab. 3). Sedimentation removes particulate bound components by gravity, leaving the dissolved fraction to be discharged or required removed in a subsequent treatment step ("Step 2" in Tab. 3). In theory, sedimentation follows Stoke's law, where settling velocity is determined by (1) the density differentiation between the particle and the water it sediments in, (2) the water's viscosity and (3) the diameter of an idealized, spherical particle (Ødegaard, 2013, p.189). In situ conditions may differ significantly from theoretical; irregular particle shapes (illustrated in App. B), turbulence, Brownian movements (random particle motions and collisions with other molecules) and viscosity variations induced by infiltrating water in tunnel wash water sedimentation basins. Despite non-idealized conditions, sedimentation has been accepted as a particle removal step.

In addition to particle removal, tunnel wash water residence time in sedimentation basins aims to fulfill minimum detergent degradation. 28 days residence time under aerobic conditions in a sedimentation basin or other treatment facility is the current minimum requirement for ensuring 60% ultimate detergent degradation to carbon dioxide and water. This requirement includes minimum 60% degradation of intermittent degradation products (Roseth and Søvik, 2006; Meland, 2012). Although the sedimentation process of tunnel wash water aims to fulfill minimum detergent degradation, the EQS and NPRA's treatment requirements do not specify compounds of interest that include the detergents' components (Miljødirektoratet et al., 2020).

# 3 Method

## 3.1 Site description

Tunnel wash water sampled from sedimentation basins in this study was collected from tunnels in Oslo and Trondheim, Norway. In Oslo, Granfosstunnelen (GfT), Smestad tunnel (SmT) and Tåsentunnelen (TT) are located along the same highway, *Norwegian National Road 706* (*Ring 3*). In Trondheim, Grillstad tunnel (GT) is located on *E6*, leading to Strindheimtunnelen (ST) on *Norwegian National Road 706*. The site description is outlined in Figure 3 below. The tunnels in Oslo (GfT, SmT, TT) and Trondheim (ST, GT) were chosen due to similarities within the two respectful cities. The similarities are mainly:

- Geometrical: two tunnel tubes of two mono-directional lanes each.
- Geographical proximity: urban environment, road salting frequency, climate, the same operational unit of the contractor and the NPRA.
- Dense and similar traffic patterns for Norwegian standards; AADT $>$ 15 000.
- Presence of closed sedimentation basins as treatment for tunnel wash water.

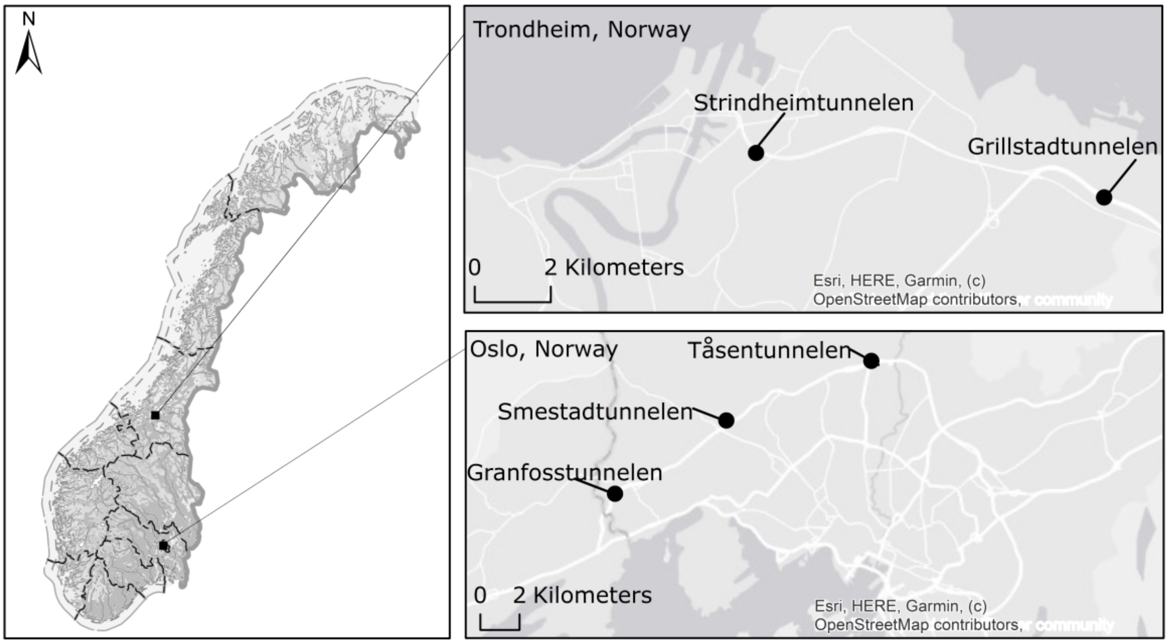


Figure 3: Tunnels sampled in this study. In Trondheim, Norway, Grillstad tunnel is located along urban highway *E6*, connected to *Norwegian National Road 706*, where Strindheimtunnelen is located. In Oslo, Granfoss-, Smestad- and Tåsentunnelen are located along the same urban highway *Norwegian National Road 150* (*Ring 3*). Highways are illustrated as white lines.

Despite similarities, geographical and operational differences between the two groups are present, as summarized in Tab. 4. The three tunnels in Oslo have separated sedimentation basins for the two tunnel tubes (Nor: tunnelløp), whilst the two tunnels in Trondheim have one sedimentation basin each, where mixed tunnel wash water from both tunnel tubes sediment. Moreover, operation and management of tunnels in the two cities, Oslo and Trondheim, are carried out by two separate departments of the same contractor. Similarly, two separate departments of the NPRA are the controlling agents. Therefore, the detergents applied and the duration between washes differ, as seen in Tab. 4.

Table 4: Summary of tunnel properties for SmT: Smestad tunnel, TT: Tåsentunnelen, GfT: Granfosstunnelen, ST: Strindheimtunnelen and GT: Grillstad tunnel. Tunnel data obtained from personal communication with the Norwegian Public Roads Administration, departments of Oslo and Trøndelag. *Water consumed pr. meter of tunnel, approximate numbers; varying with nozzle pressure, operators' evaluation, contamination levels and tunnel surface lining (Meland, 2012, Mesta Trondheim personal communication 26.03.21). **Sampled from sedimentation basin of Westbound tunnel run.

| **Tunnel data** | **SmT** | **TT** | **GfT** | **ST** | **GT** |
|---|---|---|---|---|---|
| AADT (% heavy vehicles) | | | 31 000 | 17 500 | 22 000 |
| Westbound direction | 44 060 (8%) | 22 894 (9%) | N/A | N/A | N/A |
| Eastbound direction | 22 262 (7%) | 23 046 (9%) | N/A | N/A | N/A |
| Speed limit [km/h] | 60 | 60 | 60 (70) | 60 (80) | 80 |
| Tunnel length [m] | 494 | 1338 | 2348 | 2502 | 748 |
| Washing type at sampling | Half | Half | Half | Whole | Whole |
| Inter-wash duration [days] | 36 | 40 | 36 | 28 | 28 |
| Water consumption [L/m]* | N/A | N/A | N/A | 100 | 100 |
| Detergent | Purify | Purify | Purify | Sanego | Sanego |
| Water level at sampling [m] | 1.47 | 1.42, 0.7** | 1.25 | 5 | 0.5 |
| Tunnel tube(s) sampled | West | West, East | East | West, East | West, East |
| Sampling date | 14.04.21 | 20.01.21,<br>08.02.21,<br>25.02.21,<br>15.03.21 | 14.04.21 | 25.03.21 | 26.03.21 |

The treatment plant at Smestad consists of two parallel sedimentation basins, with the purpose to remove particulate matter, followed by removal of dissolved matter in a bioretention cell. The treatment solutions were implemented as part of the most recent rehabilitation of the Smestad tunnel.

Economic restrictions for rehabilitation of the tunnel led to the construction of partly combined tunnel drainage system. Consequently, approximately one third of the drainage water from the tunnel drains to the sedimentation basin. Meaning that tunnel wash water from the Smestad tunnel is diluted to a varying degree, temporally depending on local runoff (Gremmertsen and Frost, 2015; NPRA Department Oslo, personal communication 14.01.21). Similarly, in the Tåsen tunnel, runoff is designed to make up approximately 30% of the sedimentation basins' volume during design rainfall events (NPRA Department Oslo, personal communication 18.01.21). Measured event mean concentrations (total pollution mass load divided by total water volume) of pollutants are therefore expected to fluctuate with local runoff, causing unacquainted deviations from undiluted values.

## 3.2 Sampling

Samplings were conducted while the tunnel washes were in progress or before 06.00 the same morning. Manual samplings from the tunnels' sedimentation basins were conducted using a 250 W submersible pump, with maximum 7 $m^3/h$ water flow. The pump was chosen to minimize turbulence in the water bodies to reduce re-suspension of already sedimented particles. The pump was submerged only approximately 15 cm below the water surface before turning on to minimize re-suspension further. Each sampling consisted of 300 L tunnel wash water, being pumped into 15 high-density polyethylene (HDPE) containers of 20 L each. Samples were transported from sites to the laboratory by car, i.e. approximately 30 minutes of transportation storage for the samples from Grillstad tunnel and Strindheimtunnelen (Trondheim). Samples from Oslo (Smestad tunnel, Tåsentunnelen and Granfosstunnelen) required approximately 8 hours of transportation storage. Controlled cooling of samples during transportation was not possible. Samples' temperatures were therefore slightly above outdoor air temperature, but not documented.

Practical limitations of collecting and transporting 300 L samples caused samples to be transported in HDPE containers. HDPE is non-ideal for further analyses of organic material, leading to an unclassified uncertainty in the obtained results. The containers were nevertheless chosen for transportation, because of the samples' relative short residence in the containers, under the assumption of limited leaching of organic material from the containers during the residence time. After transportation from the tunnels to the laboratory, samples was homogenized through stirring and mixed through pumping into a custom made sedimentation pilot.

The entire sampling process was repeated two times for the Tåsentunnelen (n = 2, 20.01.21 and 25.02.21), one for each tunnel run, and one time (n = 1) for each of the tunnels Smestad tunnel (14.04.21), Strindheimtunnelen (25.03.21) and Grillstad tunnel (26.03.21). Smaller control samplings (5L) of tunnel wash water were sampled twice (n = 2, 15.02.21 and 15.03.21) from the respectful runs of Tåsentunnelen after sedimentation of the previously sampled washes. Sampling from Granfosstunnelen's sedimentation basin (n = 1) was an exception, being conducted 9 hours after washing ended due to practical limitations. A previous study of tunnel wash water in Nordbytunnelen, Oslo, shows a significant decrease in rapid settable particles after 3.5 hours (Garshol, 2016). This time frame is shorter than the time in-between washing of Granfosstunnelen and the sampling conducted in this project. The time lag from washing of Granfosstunnelen and sampling leads to an unclassified uncertainty in the amount of rapid settable fraction which was not sampled. Nonetheless, the most rapid settable particles are not considered a concern for the treatment efficiency, as they are removed by sedimentation. The purpose of this 5L sampling was to be a control sample of the Smestad tunnel's tunnel wash water quality, to see if the two samples had quality parameters in the same order of magnitude, as the two tunnels were washed during the same contractor-shift. The time lag was therefore excepted.

## 3.3 Sedimentation pilot

The sampled tunnels have indoor sedimentation basins with open water surfaces and water temperatures close to outdoor air temperatures. A sedimentation pilot inside a dark, temperature-controlled room at 6 °C was used to imitate sedimentation conditions during winter in Trondheim and Oslo (Norway). The sedimentation pilot consists of four separate cylindrical columns made of poly methyl methacrylate (plexiglass), allowing visual inspection of the sedimentation process. All four columns have total height of 2 m and diameter of 0.30 m, yielding a resulting volume of 141 L for each column. The sedimentation pilot is custom-made for studying the sedimentation process of tunnel wash water at NTNU. Height and diameter combination is a compromise solution between maximizing volume within the available laboratory area. Sufficient width and height were prioritized to allow sedimentation as undisturbed as practically feasible and to imitate sedimentation basin's geometry at study sites. The sedimentation pilot's height of 2m exceeds the varying water levels in sampled basins, which varied from 0.5 to 1.47m, as seen in Tab. 4. The opposite applies to the Strindheimtunnelen sampling, where the water level of the sedimentation pilot is significantly lower than in situ water level of 5m.

To conduct analyses of duplicate sedimentation columns, $141\ \text{L} \times 2 = 282\ \text{L}$ of sample volume is needed. Additionally, 18 L of sample was collected to flush the columns with sampled tunnel wash water in advance of filling. Yielding the total $282\text{L} + 18\text{L} = 300\text{L}$ sampled tunnel wash water volume. This flushing with samples came in addition to the preceding flushing of the columns with tap water from Trondheim municipality.

Subsample volumes were collected at discrete heights of the columns using the pilot's fixed taps of stainless steel. In advance of collecting subsamples, the relevant taps were opened to allow flushing of residuals inside taps. The top, mid and bottom taps are indicated in Figure 4. The center of the respectful taps measure 185.4 cm, 107.5 cm and 11.1 cm above the interior bottom of the column, i.e., the heights are given as the water column level available. The columns were initially filled above the top tap to 2m height to sustain the water level during 30 days of sedimentation despite subsample extractions.

Another tap height in the sedimentation columns is installed below what is here referred to as the bottom tap. These lowermost taps were not used for sample extraction. Excluding the lowermost tap-height differentiates from the method of the previous master project studying sedimentation of tunnel wash water (Bjotveit, 2020). Because this tap height is 2 cm above interior bottom of the columns, it extracts mainly sediments. Furthermore, excluding the lowermost tap reduces the

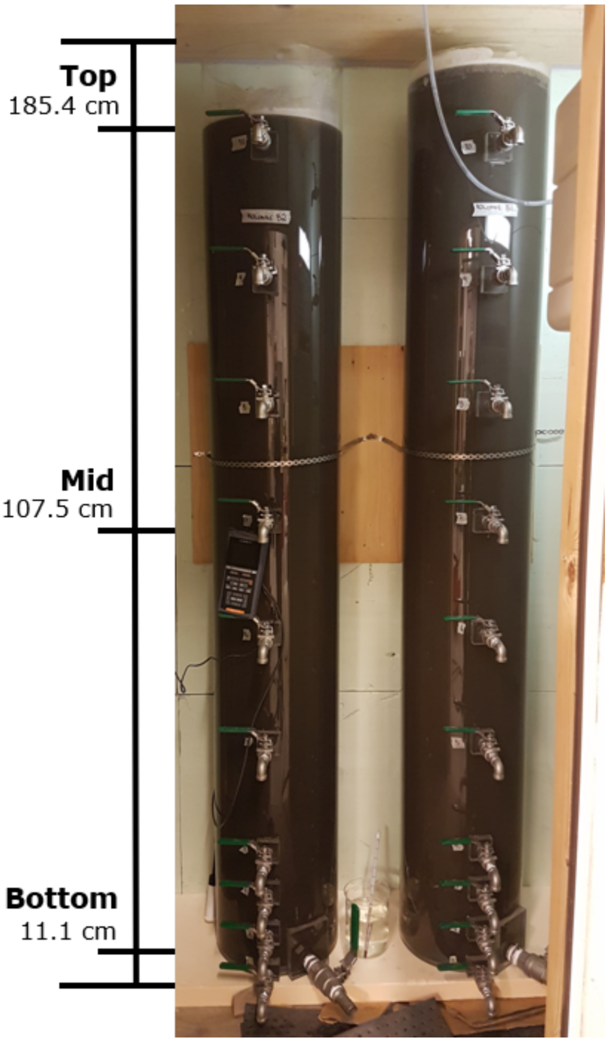


Figure 4: Sedimentation pilot setup consisting of two parallel columns with 2m height and 0.3 m diameter. The top, mid and bottom taps for extracting subsamples at discrete heights are indicated, with height measures from the interior bottom of the sedimentation pilot to the center of the taps. Illustrative water level difference between the two parallel columns.

likelihood of minor turbulence-induced re-suspension of sediments. The accumulated sediment layer interfering with the lowermost tap can be observed in Figure 21 (App. A). The sampled tunnels' sedimentation basins have separate sediment removal and deposit, making sediment samples from the lowermost centimeters of columns not representative for effluent tunnel wash water quality.

## 3.4 Standardized water quality parameters

During 30 days of sedimentation, subsamples of 500 mL to 1000 mL were collected 0, 1, 2, 4, 11, 19 and 30 days after filling the sedimentation columns. Subsample volume was adjusted to the number and type of water quality parameters being analyzed on that particular day. They were analyzed for the water quality parameters to be described below, to document what changes occurred over 30 days. The following water quality parameters were determined according to their respectful Norwegian standards: turbidity, pH, electric conductivity (EC), total solids (TS), total suspended solids (TSS) and dissolved oxygen (DO). The standards used for determining these parameters are summarized in Tab. 5 and not further described. The remaining water quality parameters are summarized in the same table 5 and discussed below.

Table 5: Temporal distribution of monitored water quality parameters during 30 days of tunnel wash water sedimentation. Norwegian Standards given where applicable. *Sampling taps in sedimentation columns as given in Figure 4. **Grillstad tunnel sample determined at days 0,1,11,19,30. EC: Electric Conductivity, DO: Dissolved Oxygen, bot.: bottom of sedimentation column, T$_{samples}$: sample temperature, IC: Ion chromatography.

| **Parameter** | **Standard** | **Tap*** | **Days** | **Remark** |
|---|---|---|---|---|
| pH [-] | *ISO 10523*, 2008 | Top, mid, bot. | 0,1,2,4,11,19,30 | Samples heated in water bath to 25°C. |
| EC [$\mu$S/cm] | *NS-ISO 7888*, 1985 | Mid | 0,1,2,4,11,19,30 | Samples heated in water bath to 25°C. |
| TS [mg/L] | *NS 4733*, 1983 | Top, bot. | 0,2,4,11,19,30** | |
| TSS [mg/L] | *NS4764*, 1980 | Top, bot. | 0, 2,4,11,19,30** | 0.45$\mu$m glass micro fibre filters used. |
| Turbidity [NTU] | *NS-ISO 7027*, 2016 | Top, mid, bot. | 0,1,2,4,11,19,30 | |
| DO [mg/L] | *ISO 5814*, 2012 | Top, mid., bot. | 0,1,2,4,11,19,30 | T$_{samples}$ = 6°C, WTW Oxi 3310 IDS digital meter used. |
| DOC [mg/L] | *ISO 8245:1999*, 1999 | Mid | 0,4,11,19,30 | Given in 3.4.1. |
| IC | *NS-EN ISO 10304-1*, 2009,<br>*NS-EN ISO 14911*, 1998 | Mid | 0,4,11,19,30 | Given in 3.4.2. |
| Heavy metals | *NS-EN ISO 17294-2*, 2016 | Top, bot. | 0,4,11,30 | Given in 3.6. |
| PSD | N/A | Top, bot. | 0,1,2,4,11,19,30 | Optical: 3.5.1,<br>Laser: 3.5.2. |

The removal efficiency or relative change of parameters listed above in Tab. 5 are later reported according to the following formula:

$$\text{Removal Efficiency}[\%] = \frac{\text{Initial concentration - effluent concentration}}{\text{Inflow concentration}} * 100 \tag{1}$$

where

Removal efficiency = Removal Efficiency of parameter
Initial concentration = Concentration of parameter before sedimentation
Effluent concentration = Concentration of parameter after sedimentation

### 3.4.1 Dissolved organic carbon

Subsamples for dissolved organic carbon (DOC) analysis were tapped from the height middle of the sedimentation pilot ("Mid" in Figure 4). This was owed to the assumption of uniform distribution of dissolved matter in the sedimentation column. Subsamples were filtered through rinsed 0.45 $\mu$m polyethersulfone filters in advance of determining dissolved organic carbon (DOC) concentrations. Filtered subsamples were conserved using 3 droplets of 21% (v/v) $H_3PO_3$ and then analyzed within one week after filtration. In advance of each analysis run, method blanks of ultra pure water (18 M$\Omega$ cm$^{-1}$, Milli-Q®) were run to check for DOC originating from the polyethersulfone filters or other equipment. Method blank DOC concentrations ranged from 0.0117 $\pm$ 0.0096 mg/L to 0.0239 $\pm$ 0.0293 mg/L. Therefore, the DOC concentrations are given with only one significant figure following the decimal point in the presented results.

DOC was determined by T. H. Næss in the Analytical Laboratory of NTNU, using Tekmar-Dohmann's *Apollo 9000 Total Organic Carbon (TOC) Analyzer*, in compliance with *ISO 8245:1999*, 1999. The method utilizes inorganic acidification of the sample, followed by gas-stripping, to drive off any inorganic carbon converted to $CO_2$. Next, high temperature combustion ($>$ 680 °C) oxidize the organic carbon in the sample. The generated $CO_2$ is then measured, resulting in a calibration curve to attain the DOC value (Wallace et al., 2002).

Previous studies of tunnel wash water quality from the Nordbytunnelen (Viken, Norway) and Smihagentunnelen (Ås, Norway) indicate that the oil, tar and PAH components in tunnel wash water mainly appear as dispersed or associated with particulate matter, i.e., included in the total organic carbon (TOC) parameter (Garshol, 2016; Roseth and Søvik, 2006). Therefore, DOC determination was chosen to filter out the particulate bound fraction of organics. The determined dissolved organic fraction is then assumed to contain the detergent components in tunnel wash water and a portion of other dissolved organic carbon species present in the DOC parameter.

### 3.4.2 Ion chromatography

Subsamples were filtered through rinsed 0.45 $\mu$m polyethersulfone filters in advance of ion chromatography (IC) analysis. Subsamples were then separately, manually diluted by a factor of 6 and 10. The resulting ion concentrations are given as a mean of the two dilutions. The analysis was conducted by T. H. Næss in the Analytical Laboratory of NTNU, utilizing the instrument 940 Professional IC Vario to detect the following analytes: $F^-$, $Cl^-$, $NO_3{}^-$, $Br^-$, $SO^{4-}$, $Na^+$, $K^+$, $Ca^{2+}$ and $Mg^{2+}$. Determination of anions and cations were in compliance with *NS-EN ISO 10304-1*, 2009 and *NS-EN ISO 14911*, 1998, respectively.

## 3.5 Particle size distribution

Two methods determined the particle size distribution of sampled tunnel wash water during sedimentation, due to their different particle detection ranges; optical particle analyses and laser diffracting. The methods are separately presented below.

### 3.5.1 Optical particle size distribution

Optical particle analyses (OPA) were conducted using PN3000 XPT Micro Particle Detector (Postnova analytics), determining both total particle number and size distributions within the range 1 to 100 $\mu$m. Prior to analysis, the instrument was flushed thoroughly with deionized water (removed for ions), then with sample, to remove possible loose, residual particles in the glass flow cell. The sample was then continuously stirred while being pumped at 4 mL/min through the inert glass flow cell of the detector until 100 pictures of the subsample were taken per OPA. The glass flow cell is continuously illuminated by LED light coupled with a sensitive charge-coupled device (CCD) detection for optical identification and characterization of particles (*pn3000-specifications - Postnova Analytics, n.d.,* ). The method yields number-weighted particle size distributions (PSDs), which are calculated based on Waddel Disk Diameter, the diameter of a disk with the same area as the particle (*Particle Measurements - NI Vision 2015 Concepts Help - National Instruments, n.d.,* ). Although particles in tunnel wash water have been indicated to contain a considerable amount of spherical particles (Bjordal, 2020), the coexistent irregular particles induce an uncertainty in the calculated Waddel disk diameters (particle sizes) (Klöckner et al., 2021). The irregularities can be observed in images from the instrument in App. B.

### 3.5.2 Particle size distribution by laser diffracting

Beckman Coulter's LS230 laser diffraction particle size analyzer was utilized to compliment the optically determined particle size distributions (PSD). The equipment was chosen to be able to identify particles over a broader size distribution than 0 to 100 $\mu$m, i.e., 0.04 to 2000 $\mu$m. Volume-weighted particle size distributions were obtained from the instrument. Previous master theses utilizing the same machine have observed that the LS230 shows affirmation for larger particle diameters in volume-weighted particle size distribution, compared to the number-weighted distribution (Strømberg, 2020; Bjotveit, 2020). The PSDs were therefore converted to number-weighted distributions using the machine's associated software. This conversion assumes symmetrical particle size distribution and spherical particles; assumptions which are not sufficiently studied for tunnel wash water. Therefore, the converted PSDs must be studied with care and is not directly comparable to the optically determined particle size distributions.

In compliance with the product manual (*Coulter LS Series Product Manual,* 2011), the particle size distributions were derived for untreated tunnel wash water. The distributions' development during 30 days of sedimentation in the pilot (described in 3.3) was studied. Beckman Coulter developed the polarization intensity differential scattering (PIDS) system to compromise for weak light scattering by the smallest particles, 0.04 $\mu$m to 0.4 $\mu$m for the LS230's range. Fractions larger than 0.4 $\mu$m are identified based on the Fraunhofer model; "*the Coulter LS230 computes the pattern of light* *scattering as a function of scattering angle for each size classification."* (Strømberg, 2020, p.9). In summary, the particle sizes are specified based on the light intensity distribution pattern.

In advance of using the Coulter LS230 and in-between sample analyses, the machine was flushed three times using deionized water (18.2 M$\Omega$ cm$^{-1}$). Deionized water was also used as background liquid, being de-bubbled before adding sample. The particle load in the background liquid was then measured, to reduce its interference with the sample. The sample was stirred to homogenize before being injected into the machine. 5 mL manual pipette was used for injecting sample until 100 mL sample was injected or when PIDS $>$ 45% and obscuration in the range 8 to 12 % were simultaneously obtained. The stop criterion was what occurred first of the two, as 100 mL is the maximum volume of the small volume module Beckmann Coulter LS230. Especially for sedimented samples, with low particle load relative to untreated tunnel wash water, lower PIDS values and obscuration than required occurred for 100 mL injection. Previous master's student on the project found varying standard deviations using the Coulter LS230 (Bjotveit, 2020). Therefore, three runs of 90 seconds duration were conducted (deviating from the default of 60 seconds). Furthermore, the present foaming agents originating from detergents led to subsamples being analyzed at minimum pump speed (30%), reducing the possible foaming.

## 3.6 Metal determination

The metal determination method consists of several steps, which are individually presented below. First, the tunnel wash water samples are filtered or separated into particulate, colloidal, and truly dissolved fractions. Next, the particulate fractions are pre-treated by UltraClave digestion. Thereby, the particles are brought into solution before metal concentrations are finally determined by Inductively coupled plasma mass spectrometry (IPC-MS).

### 3.6.1 Metal fractionation

Subsamples for metal detection were extracted from the top and bottom of two parallel sedimentation columns (as illustrated in Figure 4), for the Grillstad and Smestad tunnels. For the Tåsen and Granfoss tunnels, solely one set of metal fractions were analysed. Samples were prepared for metal analysis by separating 15 mL duplicate portions ($n = 2$) of subsamples into four fractions: unfiltered, 1.2 $\mu$m, 0.45 $\mu$m and 3 kDa. In advance of separation, subsamples were stirred to homogenize. Filters were rinsed with deionized water prior to sample filtration. Duplicate portions of subsamples were filtered through cellulose acetate filters with 1.2 $\mu$m and 0.45 $\mu$m pore sizes. The 3kDa fraction was separated by centrifuging at 4000 rpm for 40 minutes, through regenerated cellulose centrifugal filter units. The 3 kDa fraction could not be extracted from the subsamples from the Grillstad tunnel, due to delayed delivery of centrifugal units.

Metal fractionation thresholds vary, depending on the study. They are therefore defined as follows for this project: particulate ($>$ 1.2 $\mu$m), colloidal (1.2 $\mu$m to 3 kDa) and truly dissolved ($<$ 3 kDa). In addition, the 0.45 $\mu$m fraction was included as an intermediate fraction in the colloidal range, allowing broader comparison with previously published studies. The remaining fractions allow for comparisons with previous master theses written about tunnel wash water at the same project (Kowollik, 2020; Bjotveit, 2020; Bjordal, 2020), as well as previous reports on tunnel wash water quality in Norway (Garshol, 2016; Johansen and Thygesen, 2013; Aasum, 2014).

### 3.6.2 Pre-treatment of samples by UltraClave digestion

Unfiltered subsamples and subsamples filtered through 1.2 $\mu$m filters were pre-treated in an UltraClave microwave high-pressure reactor (Milestone). Combined high pressure and temperature aims to digest (decompose) organic material and other undetermined traces in the not fully understood tunnel wash water matrix (Mketo et al., 2016; Zhang and Zhou, 2020). The treatment hence brings analytes into solution, which is a prerequisite for subsequent metal detection in an inductively coupled plasma mass spectrometry (ICP-MS) instrument. Subsamples filtered through 0.45 $\mu$m and 3 kDa pore sizes were not digested with UltraClave, as material smaller than 0.45 $\mu$m is considered sufficiently dissolved to not disturb the ICP-MS instrument.

The subsamples were stirred to homogenize and then pipetted in 2 mL volume and mixed with 4 mL $HNO_3$ (50% v/v) into Milestone's associated 18 mL Teflon vessels, which were shut. The digestion tank in the UltraClave reactor was filled with 300 mL ultra pure water (5.0 M$\Omega$ cm$^{-1}$, Milli-Q®), 30 mL $HNO_3$ (65% v/v) and 2 mL $H_2SO_4$ (98% v/v). The Teflon vessels were placed inside the filled digestion tank before starting the automatic run program. Total run time for the UltraClave was 1 hour and 18 minutes for each batch, with stepwise temperature and nitrogen pressure increase from 23°C and 0 bar to 245°C and 205 bars with a constant microwave power of 1000W, as summarized in Tab. 6. Succeeding cooling time was also 1 hour and 18 minutes.

Table 6: UltraClave microwave high-pressure reactor (Milestone) program for digesting subsamples of tunnel wash water, to prepare them for metal detection. Step 2 and 8 have constant temperature (T) and nitrogen pressure (P).

| Step | Time [minutes] | T [°C] | P [bar] | Microwave power [W] |
|---|---|---|---|---|
| 1 | 5 | 50 | 45 | 1000 |
| 2 | 10 | 50 | 45 | 1000 |
| 3 | 10 | 100 | 85 | 1000 |
| 4 | 8 | 110 | 90 | 1000 |
| 5 | 15 | 190 | 155 | 1000 |
| 6 | 5 | 210 | 175 | 1000 |
| 7 | 15 | 245 | 205 | 1000 |
| 8 | 10 | 245 | 205 | 1000 |
| Cooling time | 78 | | | |

After digestion, subsamples were manually diluted by a factor of 48, that is, to 96 mL (2 mL × 48). Ultra pure water (5.0 M$\Omega$ cm$^{-1}$, Milli-Q®) was diluting agent. In practice, this was done by weighing the samples on a three decimal scale. To find the mass to dilute samples to, the following relationship was applied:

$$m = \rho \cdot V \tag{2}$$

where
$m$ is the mass of diluted sample,
$\rho$ is the combined density of diluted sample
$V$ is volume of diluted sample and

Volume was known (96 mL). $\rho$ was approximated under the assumption of all diluted samples having the same combined density of 1.0167 g/cm$^3$. The combined density was derived from combining the densities of 2 mL sample, 4 mL 0.6 M $HNO_3$ (65% v/v) and ca. 90 mL ultra pure water. The deviations from 90 mL of ultra pure water were with that neglected, as it affected the combined density to a small extent. The mass to dilute samples to on the scale, $m$, is then by Eq. 2: 96 mL $\cdot$ 1.0167 g/cm$^3$ $=$ 97.60 g. Manual filling of ultra pure water is prone to human errors, causing each subsample's final dilution volume to be controlled and corrected for deviations from 97.60 g. The volume correction was done by dividing the mass, close to 97.60 g, by the assumed constant density $\rho$ of 1.0167 g/cm$^3$. Finally, the corrected dilution factor was found by dividing the corrected volume by the volume of sample (2 mL). The individually corrected dilution factors were later applied to the metal concentrations obtained. For example, if 10 ng Metal/mL was detected, this concentration was multiplied by a corrected dilution factor of 47.9, giving metal concentration of 479 ng/mL.

### 3.6.3 Metal detection by inductively coupled plasma mass spectrometry

Inductively coupled plasma mass spectrometry (IPC-MS) enables trace metal detection in a broad range, from limits of detection (LODs) levels of 0.005 ng/mL. This range could allow metal detection in diluted samples from UltraClave digestion, as well as sedimented tunnel wash water, under the assumption of both groups having low metal concentrations, relative to untreated samples. Therefore, all metal subsamples were analysed using the ICPQQQ, Agilent 8800 (Agilent Technologies, USA) by M. Kjosbakken at SINTEF Industry, Trondheim. Analyses were carried out in compliance with *NS-EN ISO 17294-2,* 2016, to detect the following analytes: Al, As, Cr, Cd, Cu, Fe, Mn, Ni, Pb and Zn. SINTEF Industry Trondheim is certified in compliance with NE-EN ISO/IEC 17025:2017 (ISO, 2017) for As, Cu, Cd, Pb and Fe in biological material and substances of marine origin. In consequence, the metal detection in tunnel wash water deviated from *NS-EN ISO 17294-2,* 2016, point 6.13 *Matrix solution* in that the matrix solution is not validated for all analytes and their correct origin. This deviation was accepted due to NTNU's internal limitations for ICP-MS metal detection, which would not have allowed metal detection within the time span of this thesis.

### 3.6.4 Corrections for method blanks

One method blank for each of the three filter types used (1.2 $\mu$m, 0.45 $\mu$m and 3 kDa) were included in the ICP-MS analysis. The blanks were ultra pure water filtered by the same type of equipment into the same metal free tubes as the metal samples. In addition, two method blanks per run of UltraClave were included, as described above. The obtained metal concentrations were thereby corrected for three relationship, in the following order:

1. Dilution factors' deviations from 48 (Section 3.6.2),
2. Detected metal concentrations in blank samples for filters were subtracted from the detected concentrations,
3. Detected metal concentrations in blank samples for UltraClave, i.e., 4 mL $HNO_3$ 65% v/v diluted to 96 mL (97.60g) were subtracted from the detected concentrations.

The latter correction for UltraClave blanks were performed based on the two out of fourthly Teflon vessels used in the UltraClave reactor. The fourthly vessels were not checked individually for contamination history. Therefore, the average detected concentrations of the two blanks included in each UltraClave run was used as blank correction concentration. The weakness in this method is hence the unclassified likelihood of contamination variation within the vessels being neglected. Corrections were not performed for UltraClave blanks with concentrations below detection limits. In those cases where two blanks in the same run differentiated in detected concentrations, with one of the two being below detection limits, half the detection limit was used as an approximation. The average of the two was then the combined value of half the detection limit and the detected concentration of the other blank. As a last step in the corrections, the values were visually inspected and revised.

## 3.7 Statistical analysis

Two previous studies of tunnel wash water reported correlations between many of the same water quality parameters as measured in the current thesis (Meland and Rødland, 2018; Aasum, 2014). The measurements in the present study are therefore assumed to have correlations between some of the individual parameters. Principal Component Analysis (PCA) was chosen to analyse for these possible correlations and to determine those present for samples from the Grillstad tunnel (GT) and the Smestad tunnel (SmT). The method seeks to detect the strongest, theoretical trends in the measured variability of tunnel wash water quality parameters. The analysis was performed by A. Ferry, University of Lyon, intern at NTNU spring 2021. The method composes two scripts in the Open Source programming language R, given in App. E. The first script imports measured water quality parameters, to create a correlation matrix of the data. The second script uses the created correlation matrix as input to perform PCA by the integrated R function **PCA()**.

### 3.7.1 Correlation matrices

First, the input data was organized in a tabular delimited text file. Individuals of measured parameters were given as columns of the input file, such as one column for turbidity, TS, different metal concentrations and so on. Each column then had rows of the parallel variable measurements of the individuals: tunnel, day, sedimentation column, position in sedimentation columns (top, mid, bottom).

Next, the measurements were normalised, reducing the impact of outliers and differences in measurements' order of magnitude. For example, TS constituting of $10^3$ mg/L, while ion concentrations could be close to 0.1 $\mu$g/L in the same row. The correlation matrix was automatically generated by R's integrated function **cor()**. The Pearson correlation coefficient was utilized, which measures the linear dependence between two variables. This is the default setting, therefore not given as separate input. This results is a correlation matrix containing the correlation coefficients between each variable and the rest, for example the correlation coefficient of unfiltered Fe concentration in the Grillstad tunnel sample with TS, conductivity and remaining measurements.

The data set contained some missing values, i.e., DOC and IC results for day 30 of Smestad tunnel and metal concentrations below detection limits where these made up more than 5 of the data points). The "complete.obs" argument in the **cor()** function handles missing values in the data set by case-wise deletion. The correlation matrix was hence found by: **cor(x, use="complete.obs")**, where x is the input data described above.

The function **rcorr()** in the *Hmisc* package was used to compute the significance levels for all Pearson correlations. Meaning the obtained correlation coefficients from the correlation matrix are supplemented by the p-value (the probability that the correlation did not occur by chance) of the correlations for all pairs of data in the data table. The final step of preparing the correlation matrix was by re-organizing it by the custom function **flattenCorrMatrix**. The re-formatted correlation matrix then consisted of four columns: the two first being the two measurements used for the Pearson correlation test, followed by their correlation coefficients and finally their p-values. The full script, with supplementary explanations in comments are given in App. E. The final correlation matrix was saved as a tabular delimited text file, before being input to the statistical analysis itself.

### 3.7.2 Principal component analysis

The *Factoshiny* package was used to perform principal component analysis (PCA), with its built-in **PCA()** function. The PCA function aims to condense the initial input file of up to 57 columns for all measured water quality parameters, ions and metals in a space of fewest possible dimensions. These new, condense dimensions are called principal components (PC). In order find the principal components, the PCA algorithm diagonalizes the correlation matrix created in advance. Each of the up to 57 dimensions (in these measurements) will be associated with one eigen value of the correlation matrix. The dimension with the highest eigen values represent the highest theoretical percentage of variance in the variables (measurements). This is the first principal component (PC1 or dimension 1). Each following principal component has the consecutive highest obtained percentage of variations in the data, under the stipulation of it being uncorrelated with the previous principal component (orthogonal in the plane). In this study, a threshold of 80% joint variance capture in the obtained principal components was set as a threshold for acceptable representation.

The output of PCA is plotted as a score plot, displaying how variables (measurements) cluster together and how they are placed in the plane of the two first principal components. Corresponding loading plots are also produced, visualizing how strongly each measured variable's characteristics influence the obtained principal components. The principal components representing the largest theoretical explanatory variation in the measured values (two principal components in this case: PC1 and PC2). Arrows close to parallel with the first or second principal component, are well represented by that dimension. Similarly, vectors that are close to or parallel are well correlated parameters. Opposite, orthogonal vectors are not correlated. Vectors not reaching the circumference of the loading plot are not well represented by the new dimensions, meaning these correlations should be interpreted with care. In the loading plots, vectors are displayed as active or illustrative. The active vectors represent the variables included in the PCA. The illustrative variables show how they correlate with the active variables, but they are omitted when the principal components are created. In the current analysis, the following variables were excluded from the main analyses of correlations: column position (top, mid, bottom), day (0, 1, 2, 4, 11, 19 and 30) and column number (GT1, GT2, SmT1 and SmT2).

The metal concentrations' input comprised concentrations below detection limits for Cr, As, Cd and Pb for both the Grillstad tunnel and the Smestad tunnel. The data points being below detection limits were excluded if less than five ($n < 5$) measurements of the same fraction for the same tunnel were below detection limits. In the opposite end, outliers were removed through visual inspections. This was solely relevant for single values significantly overshooting the remaining data set. For example, one parallel of 1.2 $\mu$m filtered Cu for the sample from the Grillstad tunnel, which was detected to be 2061 $\mu$g/L was removed, because the remaining span was 32.4 to 45.1 $\mu$g/L. After the removal of similar outliers, no additional outliers were detected by the principal component algorithm.

Time limitations led to DOC and IC results for day 30 of the Smestad tunnel to be omitted from the PCA. This limitation was accepted due to these parameters being in dissolved fractions, lowering the likelihood of settling during sedimentation. Still, decomposition or reactions with other elements may occur, which is then omitted from the analysis.

# 4 Results and discussion

This chapter presents the evolution of tunnel wash water quality during 30 days of sedimentation, combined with the discussion of these results. First, the parameters describing particle load are discussed; turbidity, total solids, and total suspended solids. Then follows the particle size distributions determined by the methods; optical and laser diffraction. Metal fractionation and changes in metal concentrations are also presented, followed by the measured dissolved water quality parameters. The statistical analysis of all parameters summarizes the correlations between different tunnel wash water quality parameters. Lastly, unintended findings regarding asset management of sedimentation basins conclude the results.

## 4.1 Particle load parameters

The results of particle load parameters turbidity, total solids, total suspended solids, and the total number of particles have similarities in behavior, therefore presented together in this section.

### 4.1.1 Turbidity

In general, the measured turbidity of sampled tunnel wash water reduced during sedimentation. The reduction throughout 30 days of sedimentation is plotted in Figure 5 for the Grillstad (GT) and Smestad tunnels (SmT). Initial turbidity of tunnel wash water from the Grillstad tunnel was 686 ± 5 NTU, which reduced 99 % during sedimentation to 8.7 ± 0.3 NTU.

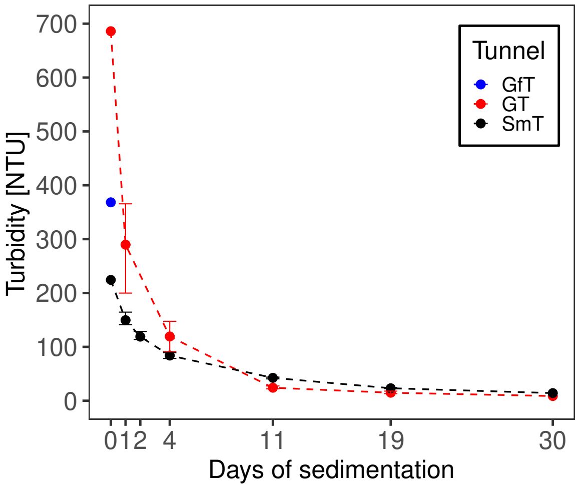


Figure 5: Turbidity decrease during sedimentation of tunnel wash water samples from the Grillstad tunnel (GT) and Smestad tunnel (SmT). Each data point averages three measurements of top, mid, and bottom of duplicate sedimentation columns (eighteen measurements per mean value). The Smestad tunnel's associated control sample from the Granfoss tunnel (GfT) given as initial turbidity.

Turbidity of tunnel wash water from the Smestad tunnel also reduced during sedimentation. The initial turbidity of 224 ± 2 NTU reduced to 14.2 ± 0.7 NTU, corresponding to a 94 % reduction. The top, mid, and bottom of the two columns with the Smestad tunnel sample followed a similar turbidity reduction pattern independent of extraction height in the columns. After four days of sedimentation, the largest absolute decrease in turbidity already occurred before the reduction rate flattened, as seen in Figure 5.

The turbidity reduction of the Smestad tunnel sample follows a similar reduction pattern as found in a previous master thesis studying the same tunnel (Bjotveit, 2020, p.11). Furthermore, the thesis of Bjotveit, 2020, found the turbidity of tunnel wash water from the Smestad tunnel to be lower than for the Bjørnegård tunnel (Bærum) and the Tåsen tunnel (Oslo) throughout sedimentation. The current results comply with the relatively low initial turbidity of the Smestad tunnel sample; the turbidity of the Grillstad tunnel and the control sample from the Granfoss tunnel both have higher initial turbidity. Although, the sampling from the Granfoss tunnel occurred approximately 9 hours after the tunnel washing event, where intermittent sedimentation likely reduced turbidity. Therefore, the *actual* initial turbidity of the Granfoss tunnel wash water is probably higher than found here, making the Smestad tunnel sample stand out more.

As stated in the introduction (Section 1), the lack of tunnel wash water quality documentation during sedimentation is evident. Therefore, comparison to external studies is challenging. The obtained documentation covers initial tunnel wash water turbidity from the Nordby tunnel (Viken, Norway) in two reports (Garshol, 2016; Aasum, 2014). The measurements have significant temporal variations up to 1000 NTU, which comply with the current measurements ranging from 224 to 686 NTU. Simultaneously, the lack of turbidity measurements highlights how the processes occurring in tunnel wash water during sedimentation are not yet fully understood.

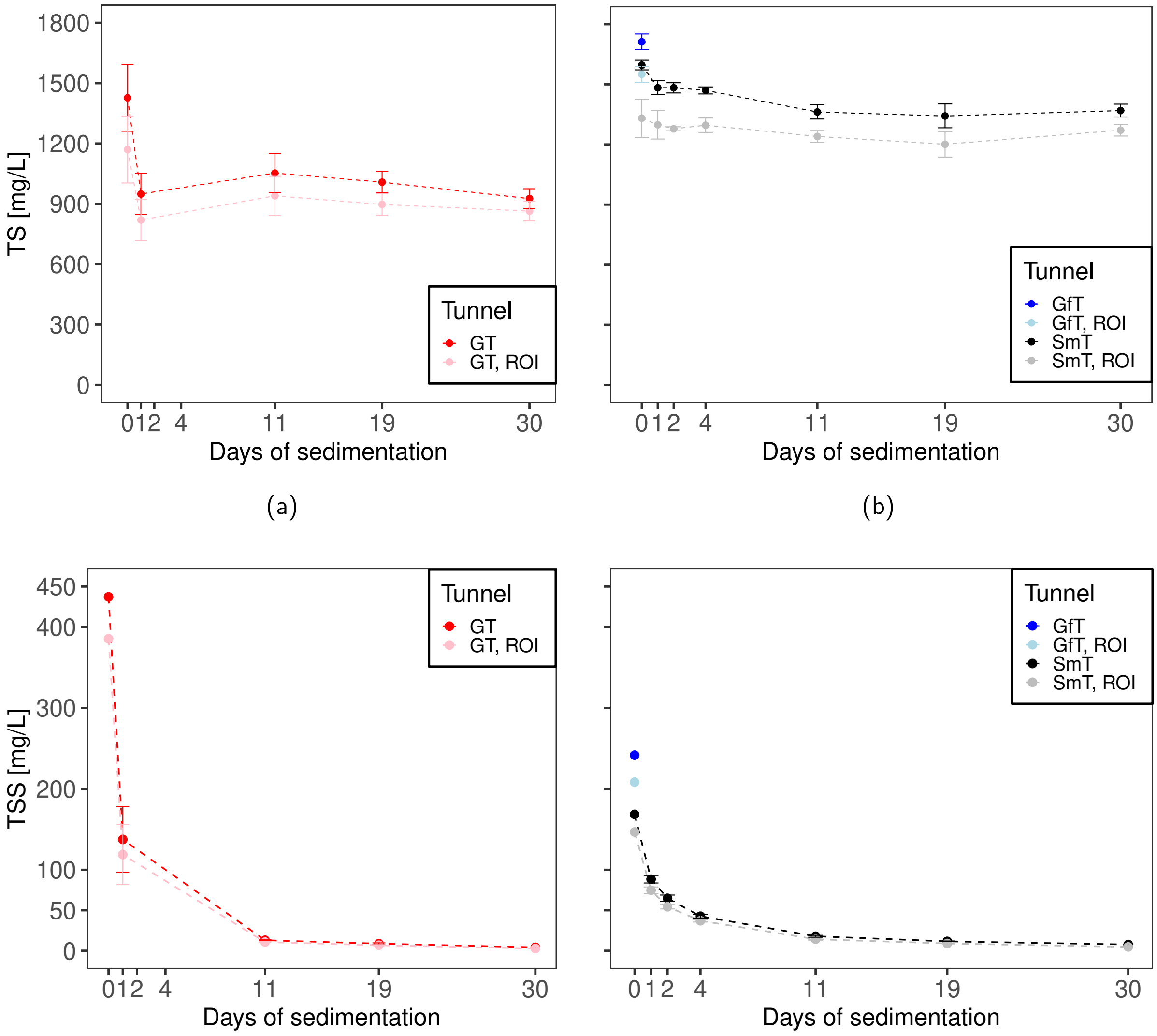
The results of particle load parameters, i.e., turbidity, total solids, total suspended solids, and total number of particles were measured for the top, mid (turbidity), and bottom of two parallel sedimentation columns with sample for the same tunnel. The four parameters are similar in that the particle load was stable in all heights of sedimentation columns with sample from the Smestad tunnel. In contrast, the parameters differentiated in the top and bottom of the sedimentation columns of the Grillstad tunnel sample the first four days of sedimentation. The top of sedimentation columns then decreased more rapidly than the bottom. The total particle number in App. B, Figure 24 exemplifies the difference between the top and bottom of the columns. After eleven days of sedimentation, the particle load parameters homogenized in column heights. The sedimented tunnel wash water quality after 30 days was in other words not affected by the initial height differences in particle load. Therefore, the parameters are presented as the mean of four values: the top and bottom values of duplicate sedimentation columns. The higher standard deviations of days 0, 1, 2, and 4 in the mean particle load parameters hence originate from this averaging of the top and bottom values.

### 4.1.2 Total solids and total suspended solids

In agreement with the turbidity results, total suspended solids (TSS; $> 0.45\ \mu$m) was efficiently removed by the sedimentation treatment, while total solids (TS; all particle diameters) moderately reduced. The initial TS concentration in tunnel wash water from the GT was 1.43 ± 0.06 g/L. TS in the same sample reduced to an average value of 0.93 ± 0.02 after sedimentation, which is a 35 % reduction. The TS and TSS reductions during sedimentation are plotted for samples from the Grillstad tunnel (GT), Smestad (SmT), and Granfoss tunnels (GfT) in Figure 6. The average residue on ignition (ROI; 550 °C ignition) made up 88 % ± 4 % of the total solids, indicating the presence of mainly inorganic solids. Simultaneously, the reduction pattern of the residue on ignition followed the total solids during sedimentation, indicating that the organic and inorganic components sedimented to a similar degree.

Average TS concentrations in the Smestad tunnel (SmT) sample reduced from an initial value of 1.69 ± 0.02 g/L to 1.37 ± 0.03 g/L during sedimentation. This corresponds to a 19 % reduction on average. The absolute reduction was in other words 16 % smaller for SmT than for GT. The residue on ignition constituted on average 89 % ± 4 % of the TS concentrations, yielding a similar indication of inorganic solids dominating, as previously seen for the Grillstad tunnel. These values are in line with previously obtained results, that found removal rates of 11 to 17 % (Bjotveit, 2020).

Starting with a concentration of 437 ± 1 mg/L, TSS in the Grillstad tunnel sample was removed below the quantification limit (LOQ) of 5 mg/L (*NS4764*, 1980), indicating removal of at least 99 %. Therefore, the average sedimented TSS concentration of 4.2 mg/L illustrated in Figure 6c can only be considered an indication of the value. For the Smestad tunnel, TSS concentration was on average reduced 95 % by sedimentation, from 169 ± 1 mg/L to 7.68 ± 0.68 mg/L. The TSS plots in Figure 6c and 6d indicate that the organic fraction of TSS is largest the first days of sedimentation before declining relative to the inorganic fraction (ROI). The Smestad sample's organic fraction decreased from 13 % initially to 7 % after 30 days. This initial decrease in the organic fractions could be owed to organic particles larger than 0.45 $\mu$m sedimenting rapidly, similar to inorganic fractions in the same size range. Opposite to what the larger absolute gaps in plot 6c indicate, the relative fraction of organics actually *increases* during sedimentation for the Grillstad sample, from 12 % initially to 27 % on average after 19 days of sedimentation (day 30 omitted; below LOQ). Opposite to the sedimentable particles larger than 0.45 $\mu$m, the persistent organics could be a mixture of micro-plastics, organic colloids or detergent components. The differentiation relies on more sophisticated determination, which is outside the scope of the current thesis.



(a) (b)

(c) (d)

Figure 6: Total suspended solids (TSS; $> 0.45\ \mu$m) and total solids (TS) decline in tunnel wash water during sedimentation. Each data point is the average of one measurement in the top and bottom of duplicate sedimentation columns (four measurements per mean value). (a): TS in the sample from the Grillstad tunnel (GT). (b): TS in the sample from the Smestad tunnel (SmT) and the corresponding control sample from the Granfoss tunnel (GfT). (c): TSS in the sample from the GT. (d): TSS in the sample from the SmT and GfT. Residue on ignition (ROI; 550 °C ignition) composing the bulk of solids in all cases.

The most considerable absolute decrease in TSS concentration appeared during the first day of sedimentation for both tunnels. The top of columns with the sample from the GT reduced from the initial value to 100 and 117 mg/L. Simultaneously, bottom concentrations were higher, with respective TSS concentrations of 164 mg/L and 169 mg/L. The sample from the SmT had less variations over the sedimentation column heights, visualized as modest standard deviations in Figure 6b. TSS concentrations became homogeneous throughout the column heights within 11 days of sedimentation (intermittent measurement not carried out for the GT), followed by more modest reduction from the same day, as plotted in Figure 6c. Significant reduction in TSS suggests that sedimentation of particles larger than the filter opening of 0.45 $\mu$m was successful. In addition, it happened rapidly during the first days of sedimentation. Co-current with TSS reduction, the TS reduction was more modest, which indicates that the particles smaller than 0.45 $\mu$m sedimented to a lesser extent. The results of optical particle detection are the relative number distribution of particle diameters (where the sum is 100 %) and the total particle number obtained during a certain measurement. Both parameters express the changes in number of present particles, which depends on the applied instrument settings. The measurement method was consistent in this study, therefore allowing comparisons between different days of sedimentation.

### 4.1.3 Total particle number

The total particle numbers declined throughout sedimentation, as visualized in Figure 7. In compliance with previously discussed particle load parameters, the total particle number decreased faster at the top than at the bottom of sedimentation columns, resulting in a higher standard deviation after averaging these values for days 0, 1, 2 and 4. The sample from the Smestad tunnel had more similar behavior over the column heights than the sample from the Grillstad tunnel, expressed as smaller standard deviations in Figure 7. However, sedimentation column 1 for the Smestad tunnel (SmT1), deviated significantly from pre- and proceeding number measurements the first day of sedimentation ("day 1"): 1193 ± 137. This measurement is in-between numbers found for 19 and 30 days of sedimentation for the same tunnel. Column 2 with sample from the same tunnel counted 16711 ± 681 during the same measurement series. Accordingly, "day 1" for SmT1 was considered an outlier and not included in Figure 7.

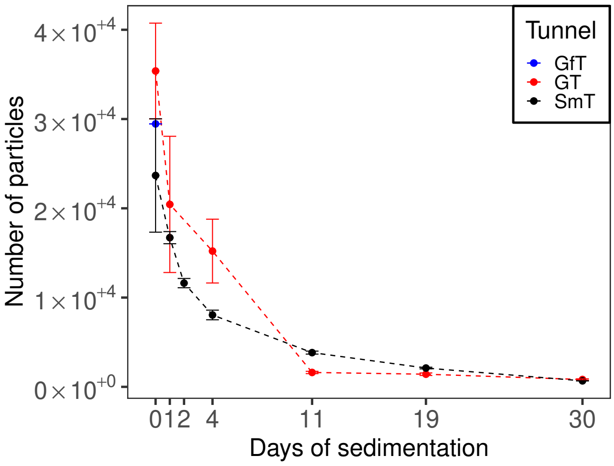


Figure 7: Decline of mean, relative number of particles with standard deviations, a result of optical particle analyses. Mean values and standard deviations were found based on four parallels with sample from the Grillstad tunnel (GT) and the Smestad tunnel (SmT). Solely the initial (day 0) analysis performed for the Granfoss tunnel (GfT), associated to the SmT.

The removal efficiency of the total number of particles was 98 % for the sample from Grillstad and 97 % for Smestad. The initial and sedimented numbers are off less interest, as they convey results obtained during a certain measurement. This number is again method dependent (described in Section 3.5.1). This high, relative reduction complies with previously discussed turbidity and TSS reduction ranging from 94 % to 99 %. The total particle numbers during sedimentation of sample from the Smestad tunnel were also studied in the thesis of Bjotveit, 2020, where the same reduction patterns for the Smestad were observed. This increases the likelihood of representative reduction patterns of the total number of particles. This parameter has not been found in external studies.

## 4.2 Particle size distributions

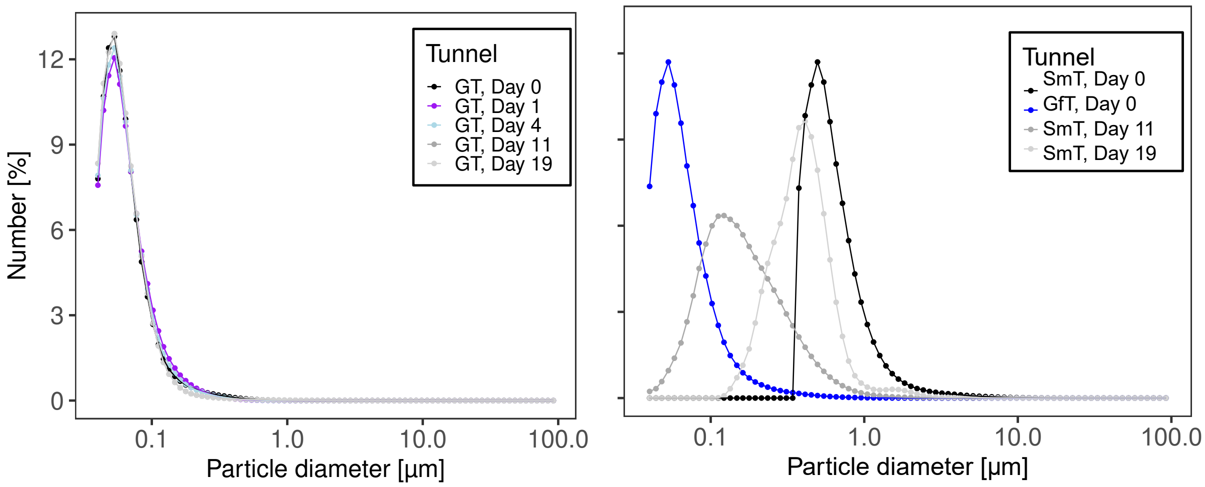
The particle size distributions of tunnel wash water samples were determined using optical particle detection and laser diffraction. Hence, the results deviate according to the methods' particle range detection, with optical particle detection in the range 1 to 100 $\mu$m and laser diffraction in the range 0.04 to 2000 $\mu$. The results of both particle detection methods are the relative number distribution of particle diameters (where the sum is 100 %). The consistency in analysis methods (Section 3.5.1 and 3.5.2) allows comparison throughout sedimentation experiments.

### 4.2.1 Optical particle size distribution

The mean number-weighted PSDs for both tunnels indicate shifts during sedimentation. The initial ("Day 0") peak of particles appears at 3 to 4 $\mu$m for the Grillstad (GT), Smestad (SmT), and the Granfoss tunnel (GfT) in Figure 8. The initial PSDs of the three tunnels have tails reaching 22 $\mu$m, meaning evident numbers of particles up to this size are present. The optical PSD of tunnel wash water from the Grillstad tunnel demonstrates a declining number of particles larger than 4 $\mu$m in diameter after 30 days of sedimentation. The decline is seen as an increasing number-weighted peak at 4 $\mu$m, and that the PDS shift towards smaller particle diameters.

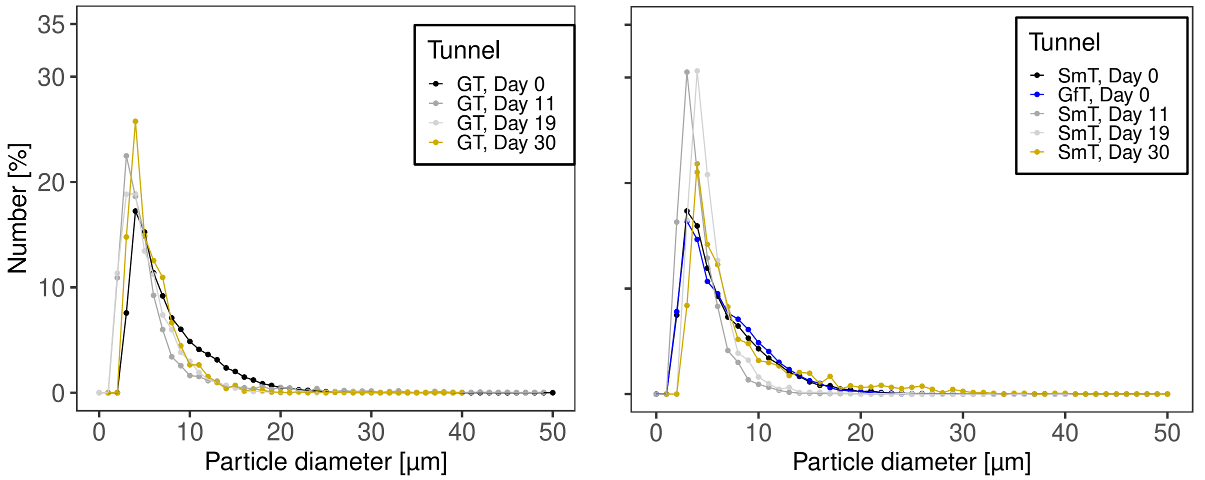
The duplicate sedimentation columns for the Grillstad and Smestad tunnels had similar behavior in the optical particle size distributions. Therefore, the mean particle size distributions of subsamples extracted from the top and bottom of the two respective tunnels' sedimentation columns are given here. The similarities between the column heights and parallels are illustrated for one, two, four and nineteen days of sedimentation of the Smestad sample in App. B, Figure 25. The first two days of sedimentation deviate from this pattern. As previously observed, rapid sedimentation of particles distinguish the top and bottom of columns the first days of sedimentation, before particle load becomes uniform.

The optical PSDs for the Smestad tunnel have less clear tendencies than the PSDs for the Grillstad tunnel. After 11 and 19 days of sedimentation, the distribution shifts towards higher number-weighted, pronounced peaks at 3 and 4 $\mu$m, respectively. The tail of larger particles simultaneously reduces. Both days demonstrate the same type of decline as mentioned for the Grillstad tunnel



(a) Laser diffraction, Grillstad tunnel

(b) Laser diffraction, Smestad tunnel and Granfosstunnel



(c) Optical PSD, Grillstad tunnel

(d) Optical PSD, Smestad tunnel and Granfoss tunnel

Figure 8: Mean particle size distributions (PSDs) determined based on four parallels of samples from the Smestad tunnel (SmT), Granfoss tunnel (GfT), and Grillstad tunnel (GT). Left: Optical PSD. Right: PSD determined by laser diffraction.

above. After 30 days of sedimentation, the PSD shifts back towards having a tail of larger particles, similar to the initial PSD ("Day 0"). This shift contradicts Stoke's law for the theoretical settling potential of particles, in addition to the reduction in TSS ($> 0.45$ $\mu$m) being stable and modestly reducing from 19 to 30 days of sedimentation. Observably, the sedimented (day 30) PSD curve of the Smestad tunnel has more noise in Figure 8d. The PSD after 30 days of sedimentation could be illustrating a method weakness. The particle load is reduced during sedimentation, leading to fewer detectable particles, allowing the influence of outliers to become pronounced. Similar patterns are observable in Bjordal, 2020, Figure 3.5, where the particles larger than 10 $\mu$m reduced from the initial distribution, but subsequent PSDs fluctuate around those diameters. The current method applied with the micro particle detector has no indicators of particle load being below detection limits, which should be further investigated if the method is re-applied.

### 4.2.2 Particle size distribution by laser diffracting

The laser diffraction method measures particle diameters up to 2000 $\mu$m. However, particles smaller than 1.0 $\mu$m dominate in numbers, making the number of larger particles negligible in the plotted PSDs in Figure 8a. Sample from the Grillstad tunnel has an initial pronounced peak of particles with a diameter of 0.053 $\mu$m, which was sustained throughout sedimentation. The PSDs throughout the 19 first days of sedimentation were also stable, illustrated by that the PSD was shifted towards an even more pronounced peak at the same diameter, while the particles larger than 0.1 $\mu$m reduced in numbers relative to the smaller particles. Similarly, the same dominating particles are seen in the initial PSD of the control sample from the Granfoss tunnel in Figure 8b.

The number weighted particle size distributions of Smestad tunnel (SmT) had an initial, pronounced peak at 0.74 $\mu$m. The particles larger than 0.12 $\mu$m then were indicated to sediment before day 11, by the shift in the PSD towards smaller particle diameters dominating in numbers. The intermittent PSDs have no clear trend (one, two, and four days of sedimentation) with shifting peaks and tails of the largest detected particle sizes. These PSDs were therefore omitted in the main results but included in App. 28. After 30 days of sedimentation, the PSDs of the Smestad tunnel could not be determined by laser diffraction. Obscuration and polarization intensity differential scattering (PIDS) were significantly below minimum requirements 8 to 12 % and 45 %, respectively. The PSD after 19 days could be influenced by reduced particle load as well, owing to its unexpected increase in peak particle diameter in Figure 8b, compared to the PSD after 11 days of sedimentation.

### 4.2.3 Comparison and discussion of both particle size distribution methods

The laser diffracting shows that particles smaller than 1 $\mu$m dominate the PSDs by number weighting. The optical particle analyses show that larger particles of 3 and 4 $\mu$m dominate the number-weighted PSDs when the particles smaller than 1.0 $\mu$m are excluded. The particle diameter range is inherited in the instrument's detection method. The control sample from the Granfoss tunnel complies with the laser diffraction derived PSDs of the Grillstad tunnel in Figure 8a and 8b. This deviates from what may be expected based on tunnel properties (as summarized in Tab. 4) and that it was a control sample from the same washing shift as for the sampling from the Smestad tunnel. Despite this unexpected similarity between the Grillstad and Granfoss tunnels initial PSDs, further evaluation of how representative these PSDs are is challenging.

Several conditions complicate comparison to previous studies. Firstly, because PSDs is rarely measured for road runoff (Selbig et al., 2016) and for tunnel dust (Klöckner et al., 2021), but evidently seldom for tunnel wash water. The same articles also point out how "*differences in sample collection, processing, analytical and reporting methods have resulted in a wide range of concentrations and* *distributions from similar sources*" (Selbig et al., 2016, p.2).

The lack of published PSD determinations in tunnel wash water is noticeable. Those published are a report briefly discussing *volume-weighted* distribution of tunnel wash water (Garshol, 2016, p.58-59) accompanied by the thesis of Bjotveit, 2020. The latter thesis reported the laser diffraction determined PSD as volume-weighted, but the optical particle detection was mainly given as number-weighted. Patterns of pronounced shifts towards affirmation for larger particles were observed for conversions between number- and volume-weighting in both the present thesis and the thesis of Bjotveit, 2020. This shift in PSDs implies that the bulk of particles are small ($< 1.0$ $\mu$m), but few, large particles ($> 100$ $\mu$m) interfere. Foaming of tunnel wash water by the pump in Beckman Coulter's LS230 laser diffraction particle size analyzer causes the need for caution when evaluating the presence of particles larger than 100 $\mu$m. They could also be foam bubbles.

Previously, particles in size ranges 20 to 50 $\mu$m and 50 to 100 $\mu$m have been found present in tunnel dust (Klöckner et al., 2021). Compared to dominant numbers of fine particles ($< 1.0$ $\mu$m) in the current thesis, the particle ranges larger than 50 $\mu$ are more likely to be removed by sweeping equipment prior to water application in tunnel washing (Klöckner et al., 2021). After sweeping, the particles left on the tunnel surface is less likely to be sampled in the current thesis' method (Section 3.2). The sampling allows the largest particles to be contained in gully pots and sand traps or settle in the sedimentation basin before sampling.

Yun et al., 2010, determined number-weighted PSDs in the range of 2 to 880 $\mu$m in road runoff, by equipment with light scattering sensor similar to the Coulter LS230 used in the current thesis and the thesis of Bjotveit, 2020. The study's derived number-weighted PDSs were converted from volume-weighted PSDs, like the method of the current thesis. Yun et al., 2010, found that particles smaller than 5 $\mu$m accounted for 80 % of the total number of particles in road runoff. At the same time, they constituted solely 12 % of the mass fraction. The results are not directly comparable to the present results, making the remarks made by Selbig et al., 2016; Klöckner et al., 2021, evident in this case as well. Still, the dominating numbers of persistent particles smaller than 1 $\mu$m in the Smestad, Grillstad, and Granfoss tunnels suggest that they sediment to a lesser extent than particles larger than 1 $\mu$m. Therefore, the mass fraction of the smallest particles ($< 1.0$ $\mu$m) would be interesting to analyze for tunnel wash water. Low mass fraction of small particles could emphasize why they sediment to a less extent, as they then would have two effects inducing reduced sedimentation efficiency, as given by Stoke's law: low relative density and small particle diameter. This was not further investigated in the present thesis but is recommended for further work, emphasized by the previously reported lack of available studies.

## 4.3 Heavy metals

Heavy metal concentrations were detected as the total concentration, which composes the fractions defined in this project, namely particulate ($> 1.2\ \mu$m), colloidal (1.2 $\mu$m to 3 kDa) and truly dissolved ($< 3$ kDa) fractions. The total removal efficiency (decline in the total metal concentration) is therefore presented first, followed by more detailed visualization of the different fractions during sedimentation.

### 4.3.1 Removal efficiency of metals during sedimentation

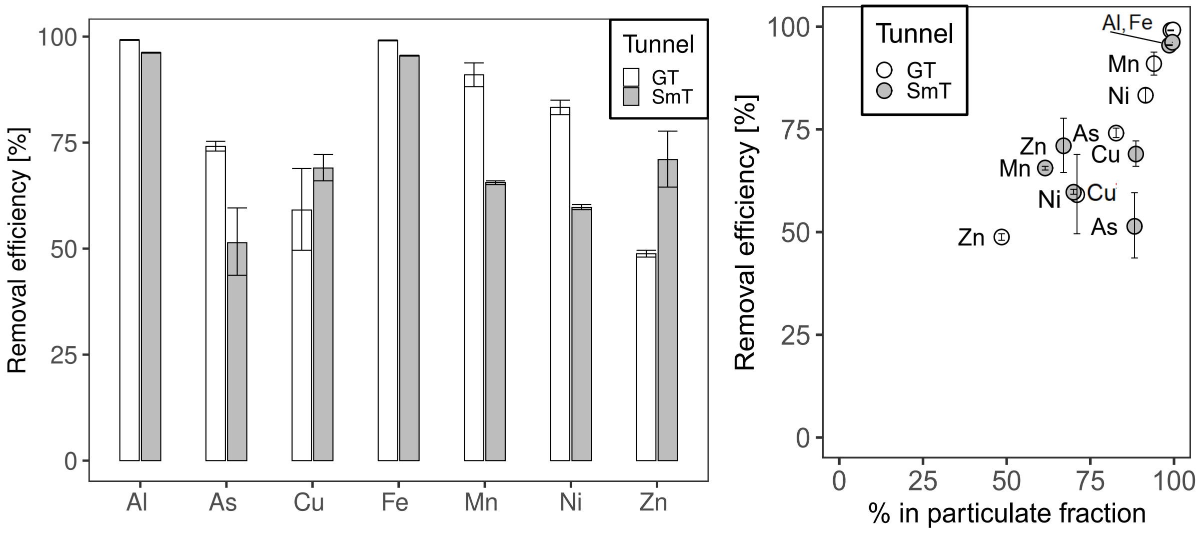
Tunnel wash water from the Grillstad tunnel had considerably higher concentrations of all total metal concentrations than the Smestad tunnel, illustrated by several plots in Figure . The tunnel wash water from the Grillstad tunnel also had higher removal efficiency of most metals during sedimentation. The exceptions were Cu and Zn, which had higher removal efficiency in the sample from the Smestad tunnel. The metal removal efficiency varies from 49 % for Zn to 99.7 % for Al. Both limits are detected in the sample from the Grillstad tunnel. Al and Fe had significant removal efficiency for samples from both tunnels (>98 %), dominated by particulate fraction reduction. Figure 9 summarizes the removal efficiency of the unfiltered (total) metal concentrations and illustrates the variance between metals and the two tunnels.

The metal concentrations indicate similar concentrations within the same sub-sample extractions from the sedimentation columns, independent of height of extraction and fraction. Especially the detected concentrations in unfiltered fractions were emphasized in this evaluation, under the assumption of particle-bound metals having larger potential to differentiate due to largest settling potential. No evident height differences were found here either. To illustrate, total Zn concentrations detected in sample from the Grillstad tunnel after 4 days of sedimentation are given in Tab. 7. The table emphasizes how the same order of magnitude of measured concentrations was found irrespective of column, parallel, and tap height (top/bottom). Zn concentrations fluctuate within the range 491 and 568 $\mu$g/L, with a mean value of 540 $\mu$g/L. In this case, the mean value is then reported with its standard deviation, based on all measurements.

Table 7: Example on metal concentrations in a similar order of magnitude within the parallels. Here given by the total zinc concentrations detected in tunnel wash water from the Grillstad tunnel, Trondheim, after four days of sedimentation. col.: Column. Column positions defined in Figure 4.

| Position in sedimentation column | Zn [$\mu$g/L], parallel 1 | Zn [$\mu$g/L], parallel 2 |
|---|---|---|
| Top, col. 1 | 545 | 568 |
| Top 2, col.2 | 561 | 491 |
| Bottom, col. 1 | 555 | 529 |
| Bottom, col. 2 | 529 | 538 |

The total removal efficiency span of 49 % to 99.7 % complies with the percentage of the particulate fraction in total metal concentrations. Similar percentages of the particulate fraction of metals in tunnel wash water varying from 40 % to 90 % have previously been reported (Meland, 2012). For instance, the mean initial concentration of particulate bound Cu in sample from the Grillstad tunnel was 67.5 ± 2.0 $\mu$g/L. The particulate fraction thus made up 71 % of the total Cu concentration of 95.1 $\mu$g/L. The particulate fraction was reduced to 4.5 ± 4.5 $\mu$g/L after 30 days of sedimentation. Despite high standard deviation relative to sedimented concentration, the particulate fraction constitutes the main removal efficiency. Simultaneously, the colloidal fraction was persistent. The truly dissolved fraction was undetermined in sample from the Grillstad tunnel (described in Section 3.6).

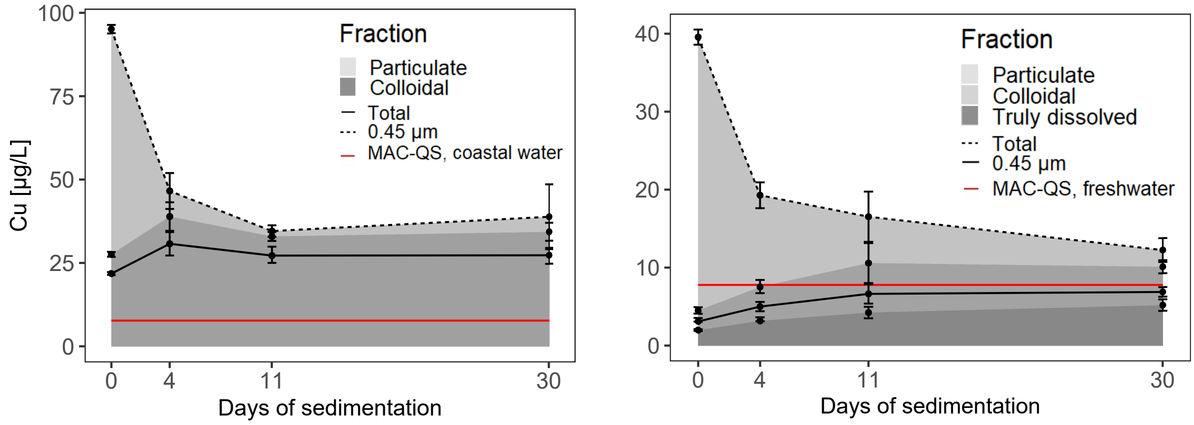


(a)

(b)

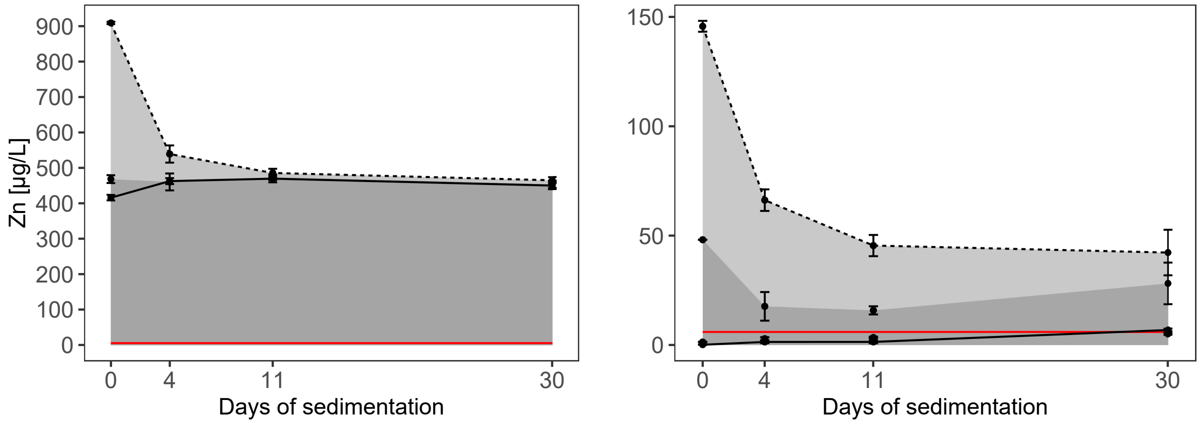
Figure 9: Graphical summary of removal efficiency for total metal concentration during sedimentation. (a): Removal efficiency of the total metal concentrations from the Grillstad tunnel (GT) and the Smestad tunnel (SmT). (b): Removal efficiency [ %] and percentage of the same metals in the particulate fraction, which are proportional for tunnel wash water sample from GT.

Similar removal patterns were observed for other metals during sedimentation, attributed to the decrease in the particulate fraction. The individual metals' decrease are presented in more detail in Figure 10, 11 and 12. The metal concentrations are given as the mean values of up to eight determinations of the top and bottom of parallel sedimentation columns. The exceptions are discussed in Section 4.3.3. The recipient threshold concentrations for short time exposure (MAC-QS) to Cu, Zn, Ni, Cr, and As are plotted along with metal concentrations. The three latter metals were detected in-between the lower and upper MAC-QS thresholds' spans; thus, both the upper and lower span thresholds are plotted in the following figures. The MAC-QS thresholds were chosen due to tunnel wash water effluent mainly appearing as point sources of pollution, where acute toxic effects may occur in the recipient. Chronic effects occur at lower concentrations (AA-QS). The Grillstad tunnel has the Trondheim fjord as recipient (coastal water), and the Smestad tunnel's recipient is the Smestaddammen (freshwater).



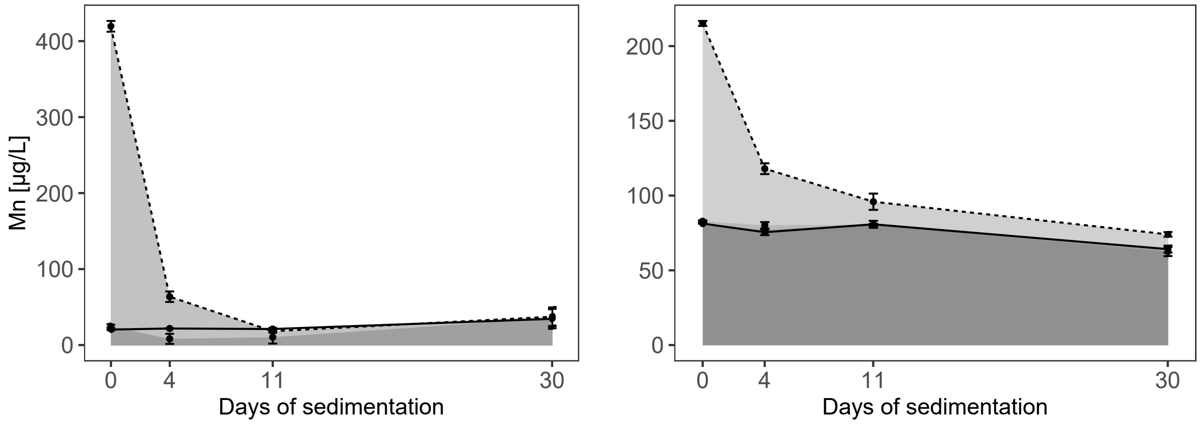
(a) Cu, Grillstad tunnel

(b) Cu, Smestad tunnel



(c) Zn, GT

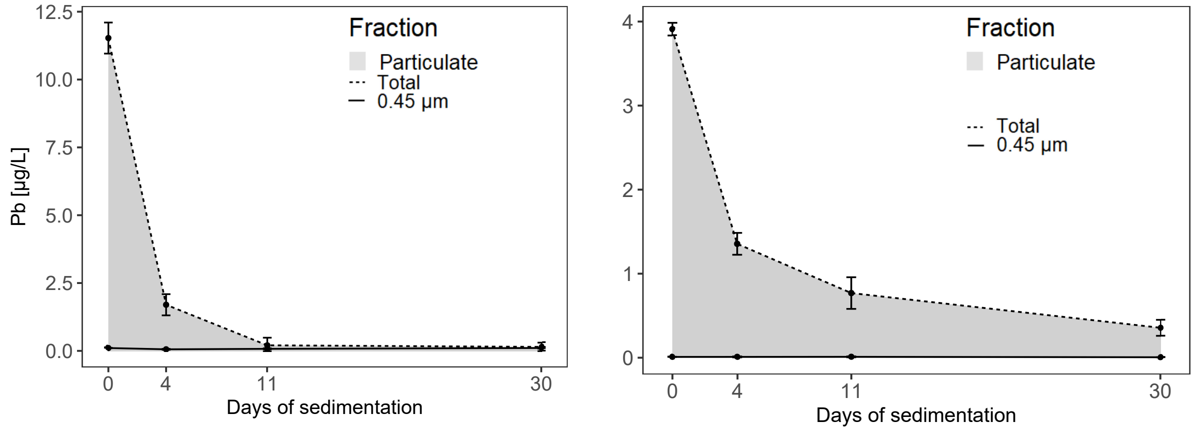
(d) Zn, SmT



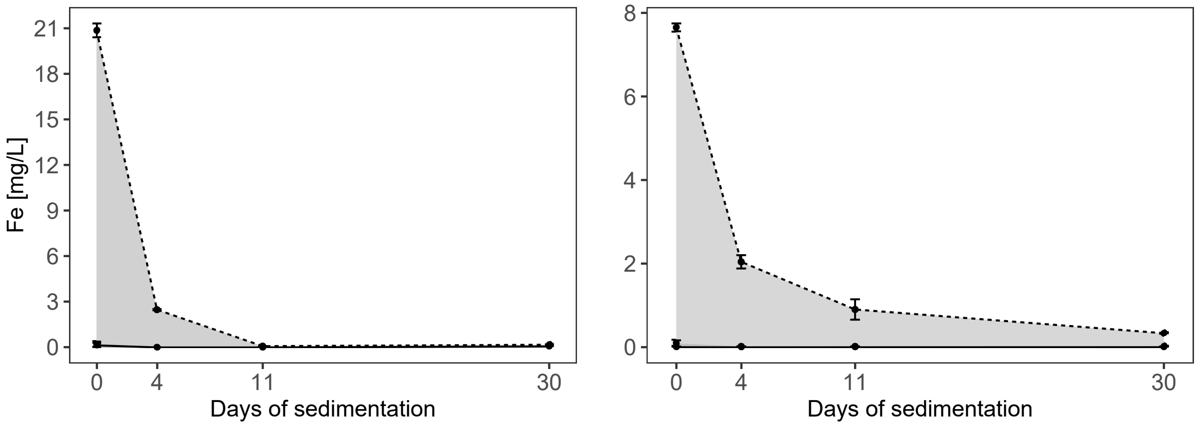
(e) Mn, GT

(f) Mn, SmT

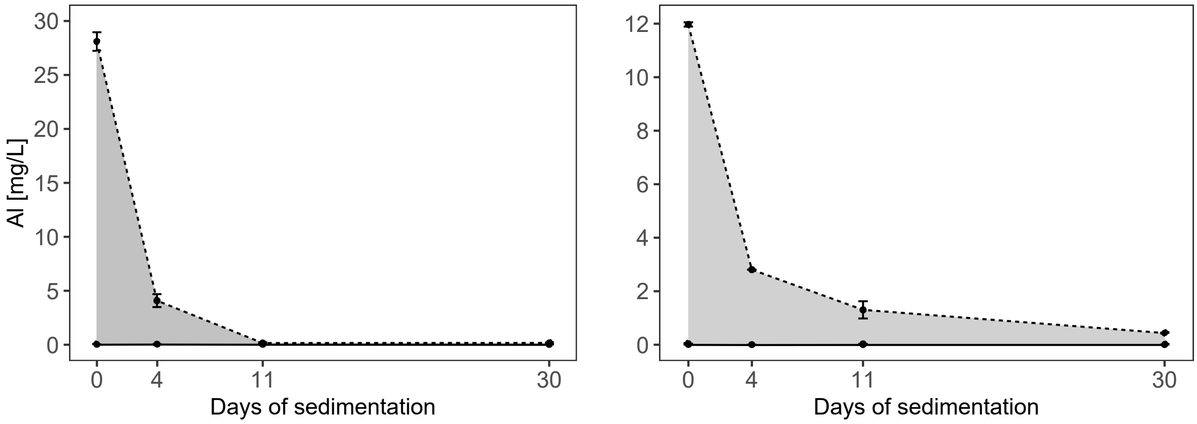
Figure 10: Copper (Cu), Zink (Zn) and Manganese (Mn) concentrations during 30 days of sedimentation in samples from Grillstad tunnel (GT, left) and Smestad tunnel (SmT, right). Please remark the difference in the y-axes between the left and right columns. The respective recipient threshold values for good water quality (MAC-QS, defined in Section 2.4) are also given for coastal water (GT) and freshwater (SmT).



(a) Pb, GT (b) Pb, SmT

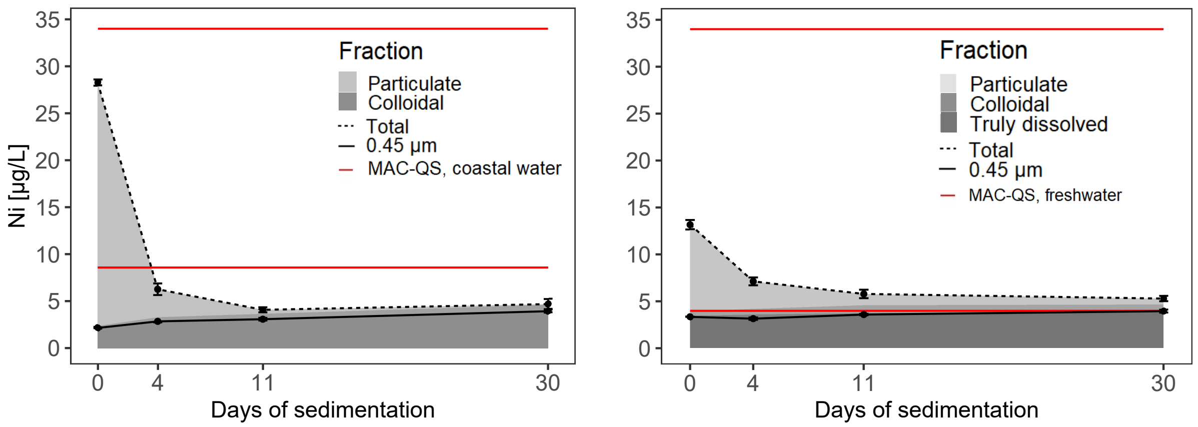


(c) Fe, GT (d) Fe, SmT

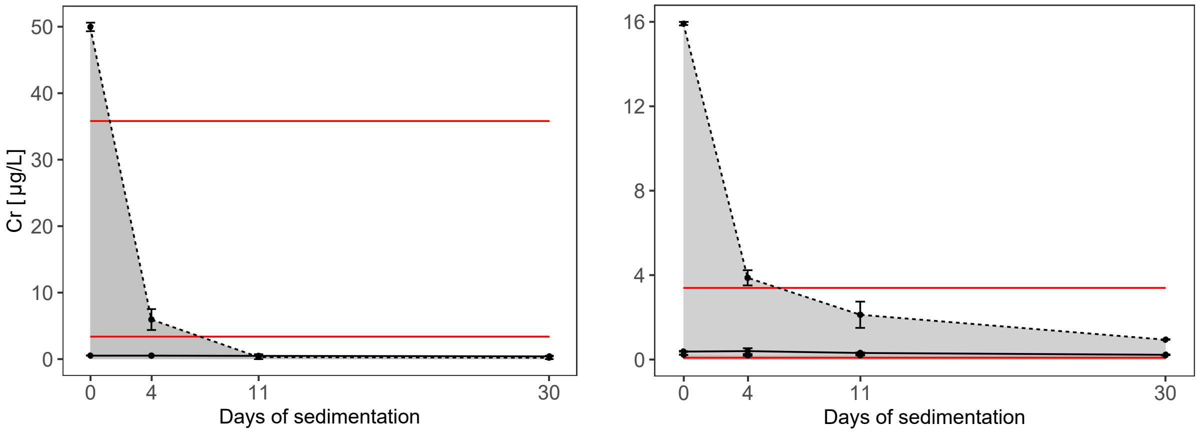


(e) Al, GT (f) Al, SmT

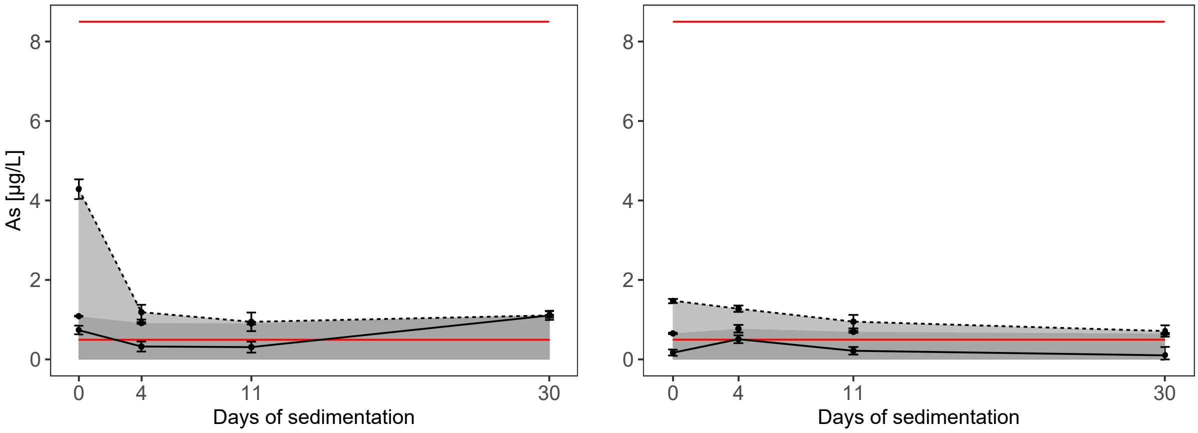
Figure 11: Lead (Pb), Iron (Fe) and Aluminium (Al) concentrations during 30 days of sedimentation in samples from Grillstad tunnel (GT, left) and Smestad tunnel (SmT, right). Please remark the difference in the y-axes between the left and right columns. Fe and Al concentrations found to be significantly higher than the rest of metals, therefore given in [mg/L].



(a) Ni, GT (b) Ni, SmT



(c) Cr, GT (d) Cr, SmT



(e) As, GT (f) As, SmT

Figure 12: Nickel (Ni), Chromium (Cr) and Arsenic (As) concentrations during 30 days of sedimentation in samples from Grillstad tunnel (GT, left) and Smestad tunnel (SmT, right). Please remark the difference in the y-axes between the left and right columns. The respective span of recipient threshold values for good water quality (MAC-QS, defined in Section 2.4) are also given for coastal water (GT) and freshwater (SmT).

### 4.3.2 Metal discharge compared to recipient concentrations

The maximum acceptable concentration quality standards (MAC-QS) for the *total* metal concentrations are exceeded for Cu and Zn for both tunnels studied. Furthermore, the bulk of the exceeded Cu and Zn concentrations are in the bio-available, dissolved form, increasing the risk of environmental harm in the recipient. The total concentrations of As in both tunnel samples and Cr in the Smestad tunnel sample exceed the lower threshold span of MAC-QS. Meaning the recipient vulnerability and dilution effects of As, Cu, Cr, and Zn should be studied to conclude on the potential environmental harm when this tunnel wash water is discharged.

The MAC-QS thresholds for Cd, Pb, and Ni are given for the dissolved concentrations in the recipient (as described in Section 2.4), where dissolved is defined as $< 0.45$ $\mu$m in Vannforskriften, 2007. Sedimented sample from Smestad tunnel has 0.45 $\mu$m fraction of Ni at the lower MAC-QS threshold span of 4 $\mu$g/L, but still below the upper threshold of 34 $\mu$g/L. Ni in the sample from the Grillstad tunnel is below a coastal recipient's MAC-QS of 8.6 to 34 $\mu$g/L. That is, below 1.3 and 1.2 $\mu$g/L in discharge from the Grillstad and Smestad tunnels, respectively. Sedimented samples from the Grillstad and Smestad tunnels had 0.45 $\mu$m Pb concentrations of 0.09 and 0.01 $\mu$g/L, respectively. With that, dissolved Pb concentrations are consistently below the MAC-QS span from 0.45 to 1.5 $\mu$g/L in discharge from both tunnels. The Pb MAC-QS threshold span goes up to 14 $\mu$g/L in both freshwater and coastal water. The MAC-QS spans were not plotted in Figure 11a and 11b, to allow visualization of Pb concentration decline during sedimentation.

The MAC-QS thresholds for Cr in coastal water vary significantly compared to the concentrations detected in this study. The MAC-QS span is from 3.40 to 35.8 $\mu$g/L, compared to reduction from $50.0 \pm 0.9$ $\mu$g/L to $0.33 \pm 0.01$ $\mu$g/L for Grillstad tunnel. This span is regardless above the detected concentrations, as visualized in 12d. In freshwater recipients, the upper MAC-QS threshold is 3.4 $\mu$g/L, which the total concentration of sample from Smestad tunnel is well below.

The MAC-QS thresholds indicated are based on short-duration exposure, i.e., the events of emptying sedimentation basins. Furthermore, the MAC-QS are given as the *recipient* concentrations, not for the discharge. In addition to the short time exposure occurring at regular discharge intervals, the chronic recipient effects are also of concern. The thresholds for chronic exposure induced toxic effects (AA-QS) are generally equal or lower for compounds of interest (Miljødirektoratet, 2018). A previous study of tunnel wash water applied ten times the AA-QS values as an approximate effluent threshold to allow better comparison of the discharge concentrations to diluted recipient concentrations (Meland and Rødland, 2018). In the present thesis, the recipients' dilution effects are not studied, neither is the toxicity of tunnel wash water effluent as a whole. Hence, this approximation was not utilized.

### 4.3.3 Metal concentrations corrected for method blanks

Metal concentrations were detected in samples and method blanks. The metal concentrations detected in filter blanks (filtered ultra pure water) and UltraClave blanks (4 mL 0.6 M $HNO_3$, 65% v/v, diluted to 96 mL) are summarized in Tab. 8. The correction for filter blanks in the 1.2 $\mu$m fraction gave 7.54 and 0.581 $\mu$g/L correction for Zn and As, respectively. The same correction was found unfeasible for Al and Fe because both analytes had higher blank values than multiple subsamples filtered through the same 1.2 $\mu$m filters (n = 34 for Al, n = 35 for Fe). Meaning Al and Fe concentrations were not corrected for filter blanks.

In summary, several metals appear after filtration in all fractions. The inconsistency in detected filter blank concentrations indicates a method weakness, which could have been improved by including a minimum of three filter blanks of each fraction to evaluate the variations better. The sequence of Zn in all three filter blanks and several detected metals in the 3 kDa filter blanks underlines the need for more blank measurements. The weakness was not further studied in the present thesis, but it should be before re-application of the method.

Table 8: Detected method blanks for metal fractionation; filtration of ultra pure water through 1.2 $\mu$, 04$\mu$m, and 3 kDa filters, and blanks from runs of UltraClave digestion (4 mL 0.6 M $HNO_3$, 65% v/v). < LOD: below limit of detection. *Elevated method blank concentrations higher than several samples (n = 34 for Al, n = 35 for Fe), therefore excluded from corrections.

| | **Method blanks, [$\mu$g/L]** | | | |
|---|---|---|---|---|
| Analyte | 1.2 $\mu$m | 0.45 $\mu$m | 3 kDa | UltraClave |
| Al | 80.6* | < LOD | 1.60 | 17.1*, 0.228 |
| As | 0.581 | < LOD | < LOD | < LOD |
| Cr | < LOD | < LOD | < LOD | < LOD to 1.74 |
| Cd | < LOD | < LOD | < LOD | < LOD |
| Cu | < LOD | 0.133 | 0.0139 | < LOD to 0.625 |
| Fe | 49.2* | < LOD | < LOD | 11.6*, 0.170 |
| Mn | < LOD | < LOD | 0.218 | < LOD to 6.09 |
| Ni | < LOD | < LOD | 0.105 | < LOD to 0.331 |
| Pb | < LOD | < LOD | < LOD | < LOD |
| Zn | 7.54 | 2.49 | 0.277 | < LOD to 7.02 |

The UltraClave method blanks gave ambiguous results. Three consecutive runs of UltraClave differentiated in blank metal concentration. The blanks from the first run had elevated average concentrations of Al and Fe, respectively 17.1 $\mu$g/L and 11.6 $\mu$g/L. These are marked in Tab. 8. In contrast, the average values of the same analytes were 0.228 and 0.170 $\mu$g/L in the following two runs of analyses. Therefore, the vessels were likely contaminated before use in the first run, beyond what could be removed with acidification and rinsing with ultra pure water. The contamination was not persistent in the following two runs. Accordingly, the contaminated samples of the first run were corrected for elevated Al and Fe contamination levels: day 0, 11, and 30 for the Grillstad tunnel. In addition, all subsamples from day 4 of sedimentation for the Smestad tunnel were corrected for the same method blanks. Although the corrections for Al and Fe were elevated compared to remaining blank analytes, they were low compared to detected sample concentrations in mg/L order of magnitude, i.e., thousand times higher sample concentrations (discussed in Section 4.3.1). The rest of the subsamples from the Grillstad, Smestad, and Granfoss tunnel were corrected for the respective, lower blank concentrations given in Tab. 8.

The UltraClave run with samples of the Smestad tunnel day 30 gave one out of two method blanks with concentrations overshooting those detected in vessels with samples. Therefore, this blank was considered an outlier and excluded from corrections, which were then based on the remaining blank.

### 4.3.4 Metal fractions below detection limits

Visual inspection of detected metal concentrations was after corrections for method blanks. The process excluded analytes with less than five detected concentrations (n $<$ 5) in further analyses. A similar number of data points for setting exclusions were not relevant for other analytes, as they were consistently present above detection limits. The exclusions listed below apply to both samples from the Grillstad tunnel and the Smestad tunnel:

1. **Cd**: below detection limits in total and 1.2 $\mu$m fractions, therefore these two fractions were excluded from further analyses.
2. **Pb** and **Cr**: frequently absent in 1.2 $\mu$m fraction, therefore this fraction was excluded from further analyses.
3. **As**: below detection limits for all measurements in the 3 kDa fraction, with one exception for GT day 30, which was excluded from further analyses. This indicate that As appeared mainly as colloidal and particulate.

Detected concentrations of 1.2 $\mu$m fraction Pb consistently had subsamples below detection limits. The unfiltered samples had Pb concentrations above detection limits, also after dilution, of 11.5 $\pm$ 0.6 $\mu$g/L in sample from the Grillstad tunnel and 3.9 $\pm$ 0.1 $\mu$g/L in the Smestad tunnel sample. The total Pb concentration is hence high enough to be detected after 48 times dilution, while the 1.2 $\mu$m fraction is not.

0.45 $\mu$m and 3 kDa fractions of Pb were significantly lower than the total concentrations. The 3 kDa fraction was reported above detection limits in three subsamples from the Smestad tunnel (day 4 and day 11), but insufficient data points (n $<$ 5) led to this fraction being excluded from plots in Figure 11a and 11b. These samples were undiluted, but still below detection limits in most measurements, indicating that Pb is mainly absent in the truly dissolved fraction while appearing as particulate bound in the sampled tunnel wash water. The latter complies with previous results in the thesis of Kowollik, 2020. The truly dissolved fraction was differently reported by Kowollik, 2020, where it appeared in modest concentrations in tunnel wash water from Smestad tunnel and the Bjørnegård tunnel (Bærum, Norway). In compliance with current detection, Kowollik, 2020 also found dissolved Pb to be absent in tunnel wash water from Tåsen tunnel (Oslo, Norway).

ICP-MS did not detect fractions of Cd treated by UltraClave digestion, i.e., the total and 1.2 $\mu$m fractions. Simultaneously, Cd concentrations were below detection limits of 0.005 $\mu$g/L in all blanks for both UltraClave digestion and filters. Similar dilution effect as for Pb is therefore likely for Cd. When relatively low concentrations are diluted 48 times, this could place diluted concentrations below detection limits. Although, the undiluted samples may not actually be, as was indicated in the thesis by Kowollik, 2020. The same study reported Cd concentrations in the particulate, colloidal and dissolved fractions, defined as in this thesis as well. The concentrations were nonetheless close to detection limits, reducing from initial, total concentrations around 0.09 $\mu$g/L to a range from 0.01 to 0.04 $\mu$g/L. The Cd concentrations reported in Kowollik, 2020, and the present thesis were thereby below previous reported total concentrations in the range 0.21 to 11 $\mu$g/L (Meland and Rødland, 2018) and 0.46 to 0.67 $\mu$g/L (Aasum, 2014; Meland, 2012).

The 0.45 $\mu$m fraction of Cd in tunnel wash water from the Grillstad tunnel had a stable detected concentration of 0.1 $\mu$g/L, which modestly reduced to 0.09 $\mu$g/L after sedimentation (only one significant figure given due to the uncertainty in the results). The effluent 3 kDa fraction was also detected to be 0.09 $\mu$g/L in the same sample. In sample from the Smestad tunnel, the 0.45 $\mu$m fraction of Cd increased from 0.006 to 0.01 $\mu$g/L during sedimentation. Simultaneously, the 3 kDa fraction followed the same pattern, slightly increasing from 0.006 to 0.01 $\mu$g/L. Accordingly, the 0.45 $\mu$m fraction of Pb is consistently below the MAC-QS span from 0.45 to 1.5 $\mu$g/L in discharge from both tunnels.

The common feature found for Cd, Pb, and Cr (1. and 2. listed above) was the absence in sample fractions digested in UltraClave. Simultaneously, the same metals were detected in the undiluted fractions of 0.45 $\mu$m and 3 kDa. Especially when the 1.2 $\mu$m fraction was diluted after digestion, its concentrations ended up below detection limits. The total, unfiltered metal fraction is expected to include higher concentrations than the filtered fractions, therefore detected in most cases. The dilution was performed to include all sample residual from UltraClave vessels' interior walls and maintain consistency in the ICP-MS analysis matrix. On the other hand, the method has now been indicated to have limitations for metal fractions of relatively low concentrations. This could be improved by diluting the samples by a factor lower than 48.

## 4.4 Parameters describing dissolved quality

Sedimentation is expected to reduce particles larger than the dissolved fractions, that is, particles larger than 0.45 $\mu$m or 3 kDa, depending on the study. Still, several processes simultaneously occur in the tunnel wash water during the sedimentation residence time. The following parameters are all describing the presence of particles smaller than 0.45 $\mu$m , therefore grouped as dissolved parameters. The two first parameters, dissolved oxygen, and organic carbon, both indicate biological degradation occurring during sedimentation. These parameters are followed by ph, electric conductivity, and ions.

### 4.4.1 Dissolved oxygen

The average measured dissolved oxygen (DO) concentration of untreated tunnel wash water was 11.94 ± 0.02 mg/L for Grillstad tunnel, at 7°C. Similarly, the initial DO concentration was 11.97 ± 0.47 mg/L for the Smestad tunnel. Theoretical maximum saturation concentration of oxygen in freshwater at 7°C is 12.14 mg/L (Benson and Krause, 1980), meaning that the measured initial DO concentrations were high; 98 % and 99 % of theoretical maximum, respectively. Another study measured a similar DO concentration in untreated tunnel wash water from the Nordby tunnel (Viken, Norway) of 11.7 mg/L (Garshol, 2016, p. 78). Lower DO concentrations of 8 to 9 mg/L have been measured in untreated tunnel wash water from the Bjørnegård tunnel (Bærum, Norway) and the Tåsen tunnel (Oslo, Norway) (Bjotveit, 2020, p. 17).

DO concentrations in the top of sedimentation columns declined throughout the 30 days, as plotted in Figure 14. Sedimentation columns with tunnel wash water from the Grillstad tunnel reached final DO concentrations of 4.78 mg/L and 5.21 mg/L, respectively. For the Smestad tunnel, DO concentrations had more modest reductions during sedimentation, to final concentrations of 7.20 mg/L and 7.56 mg/L in the top of columns. Accordingly, the DO concentrations reduced 56 to 60 % in sample from the Grillstad tunnel and 37 to 40 % for the Smestad tunnel.

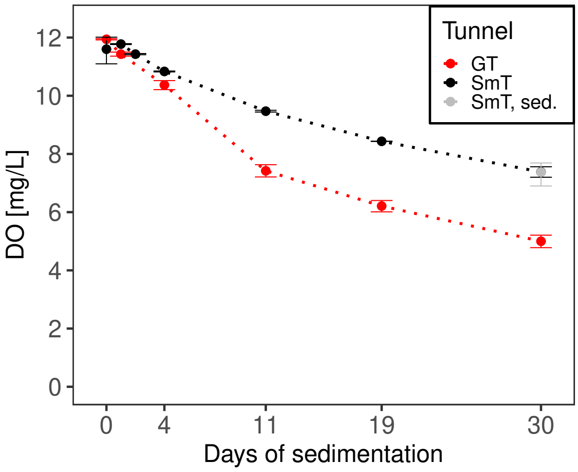


Figure 13: Declining dissolved oxygen (DO) concentrations in tunnel wash water from the Grillstad tunnel (GT) and Smestad tunnel (SmT) during sedimentation. SmT sed. is the mean of DO measurement in sediments from SmT.

Long-term sedimentation can lead to oxygen deficiency in tunnel wash water discharge, due to oxygen consumption during aerobic degradation. This was seen in the report on the Espa tunnel (Viken, Norway) by Roseth, 2013, where the sampled tunnel wash water contained 2.4 mg $O_2$/L at 4°C after 4 weeks of storage, without oxygen feed. Bjotveit, 2020, previously measured DO concentrations in the lowermost 1 cm of tunnel wash water in the sedimentation columns studied in the present thesis as well (below what is referred to as "bottom" tap here). The same thesis found lower, decreasing DO concentrations in both bottom and top of columns throughout sedimentation, than for the Grillstad tunnel (GT) and Smestad tunnel (SmT). After 35 and 44 days, the bottom DO concentrations reached anaerobic levels (DO $<$ 1 mg/L) for two tunnels. The top layer had final DO concentrations of 2 to 4 mg/L at 7°C. Although these values are lower than the values 30 days of sedimentation yielded for GT and SmT, concentrations were not measured after the same time of sedimentation, meaning direct comparison is unfeasible.

Another study of the sedimentation pond receiving tunnel wash water from the Nordby tunnel (Viken, Norway) found DO concentrations in the range 4 to 6 mg/L. This value was then for a mix of sedimented and untreated tunnel wash water, with free water surface (Meland et al., 2010). Direct comparison to these values is not possible. However, they indicate that the free water surface sedimentation of tunnel wash water from the Grillstad tunnel and the Smestad tunnel allows oxygen levels of similar order of magnitude as outdoor sedimentation ponds.

DO concentrations in extracted subsamples of the mid and bottom of columns were repeatedly measured 0.0 to 0.5 mg/L higher than measurements in the top of columns. The free water surfaces at the top of columns are assumed to allow better oxygen feed. Therefore, the DO concentrations contradict what may be expected. This contradiction may be the result of samples being disturbed during extraction from the pilot's taps. The extraction method from taps to containers was varied and tested throughout the available time frame, but the extraction likely aerates the sample regardless. Similarly, Bjotveit, 2020, remarked how "*additional air bubbles must be considered as a general source of error*" for DO concentrations in samples collected from the same taps. Therefore, the DO measurements from mid and bottom taps were not plotted in Figure 14. Nevertheless, the consistency in DO concentrations in top, mid, and bottom could indicate aerobic conditions throughout the sedimentation columns' height. The aeration is unlikely to disturb the sample to such an extent that it shifts from anaerobe (DO$<$1 mg/L) conditions to aerobic (DO$>$1 mg/L). In that sense, the slightly elevated DO concentrations in the mid and bottom of sedimentation columns indicate similar oxygen supply and consumption, independent of depth from the free water surface.

The DO concentrations of the sediment layers in the bottom of columns with sample from Smestad tunnel were measured to check if similar, anaerobic conditions apply here, as indicated by Bjotveit, 2020. The extraction was done with the minimum tap opening possible to reduce the discussed aeration effect. Measured DO concentrations of the Smestad tunnel's sediments show similar values as sedimented tunnel wash water ("Day 30") of 7.37 ± 0.42 mg/L, as indicated in Figure 14. Therefore, the joint impression is that aerobic conditions were sustained throughout sedimentation for the SmT and GT, independent of water level, with concurrent decreasing DO concentrations.

DO measurements were not corrected for salinity, due chloride and sodium concentration being detected under 1 g/L for sampled tunnels during the sedimentation process (*Operating manual Oxi 3310 IDS,* 2014, p.24). The ion concentrations are later discussed in Section 4.4.4. Concentrations of other salts that could interfere with the DO measurements were not further investigated.

### 4.4.2 Dissolved organic carbon

Dissolved organic carbon (DOC) reduced at a slower rate than the dissolved oxygen discussed above. For sample from the Grillstad tunnel, average DOC concentration reduced from an initial value of 29.5 ± 0.5 mg/L to 25.3 ± 0.4 mg/L during 30 days of sedimentation, as illustrated in Figure 14. This corresponds to modest DOC concentration reductions of respectively 14 % and 15 %. The initial value and modest reduction pattern of DOC concentration in sample from the the Grillstad tunnel harmonize with 29 mg/L and 24.5 mg/L found for the Bjørnegård tunnel (Bærum, Norway) and the Tåsen tunnel (Oslo, Norway) earlier in the thesis of Bjordal, 2020.

The tunnel wash water from the Smestad tunnel was found to have an initial DOC concentration of 10.91 ± 0.091 mg/L. The DOC concentration reduced to 8.7 ± 0.3 after 19 days of sedimentation, corresponding to 20 % reduction. The final DOC concentration after 30 days is not available, due to technical limitations in the laboratory. The initial DOC concentration comply with previous value of 10.2 mg/L measured for the same tunnel (Bjotveit, 2020). However, the DOC concentrations in samples from the Smestad tunnel are in the lower range of the previous reported median concentration of 34 mg/L for 34 Norwegian tunnels studied (Meland and Rødland, 2018). However, the same study found significant variations in DOC concentrations. Aasum, 2014, reported untreated DOC concentrations significantly higher, ranging from 140 to 170 mg/L, overshooting concentrations found in the Grillstad tunnel as well. This could be explained by several spatial and temporal variations in the sampling. Most pronounced is likely the recent shift in contractor performing the tunnel washes in the Oslo area. Furthermore, detergent types also differ.

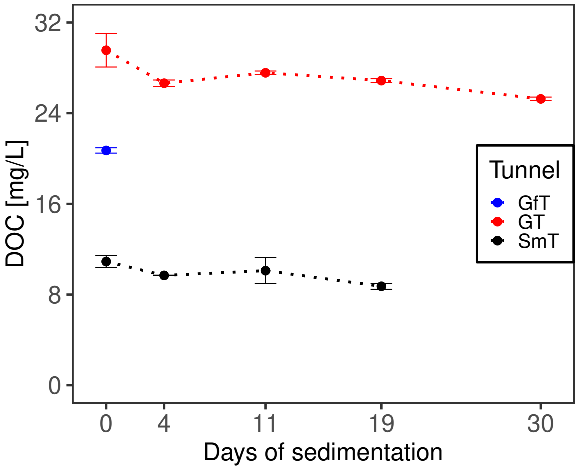


Figure 14: Dissolved organic carbon (DOC) concentration in tunnel wash water from the Grillstad tunnel (GT) and Smestad tunnel (SmT) during sedimentation. DOC concentration of day 30 for the SmT not available.

The Bjørnegård, Tåsen, and Smestad tunnels are washed by the same operational contractor division, which distinguish them from the Grillstad tunnel. Consequently, the DOC concentration that could be attributed to detergents was therefore expected to be of similar order of magnitude for similar washing procedures. When they are not, i.e., the Smestad tunnel stands out with lower DOC concentrations, the need for finding a better indication of initial detergent concentrations in tunnel wash water samples is present.

Oslo municipality, where the Tåsen tunnel (TT), Smestad tunnel (SmT) and Granfoss tunnel (GfT) are located, has no regulations requiring contractors to report detergent consumption. Consequently, initial detergent concentrations in tunnel wash water from these tunnels could not be determined. The NPRA could give a rough estimate of 70 L Purify Infravask detergent consumption per tunnel run for the Granfoss tunnel (O. Kjosbakken, NPRA Oslo, personal communication 15.02.21). Therefore, the water consumption and consequently the detergent concentration are still not available for tunnels in the Oslo area.

The Grillstad and Strindheim tunnels are washed by another division of the same contractor, using Sanego Solvynol detergent. Personal communication with the contractor's department in Trondheim gave 470 to 490 L detergent estimations and approximately 550 $m^3$ tap water consumption for washing both runs of the Strindheim tunnel 23.-25.03.2021. This corresponds to a final detergent concentration reaching the sedimentation basin of roughly $470L/(550\times1000L) = 0.085$ % to $490/(550\times1000L) = 0.089$ %. The uncertain accuracy calls for the concentration to be given with one significant figure: 0.09 %. The roughly estimated detergent and water consumption used to wash the Grillstad tunnel 25.-26.03.2021 were 90L and 100 $m^3$, respectively. The resulting estimated detergent concentration is then 0.09 %. Estimates of detergent concentrations in the two tunnels are thereby in the same order of magnitude but below previously reported concentrations in the range 0.2 % to 0.5 % (Meland, 2012).

The uncertainty in estimated initial detergent concentrations leads to limited value of comparing them to measured DOC concentrations. Furthermore, the density of detergents are not given by producers (App. D and C), neither is the full declaration of components. Hence, the volumetric percentages of detergents were not converted to concentrations, due to the inherited uncertainty.

The modest DOC concentration decreases found during sedimentation could be attributed to particle adhesion, as an additional or alternative effect to degradation. Surfactants in detergents have sorption affinity for particles, making them frequently appearing in sediments (Ying, 2006). The prerequisite of adhesion-induced reduction, namely particle reduction, was found simultaneously with DOC reduction for the Grillstad and Smestad tunnels (particle load previously discussed in Section 4.1). The DOC reduction during sedimentation of the two tunnels' samples still differentiated. This could be explained by the higher number of particles initially in the Grillstad sample (Figure 7), i.e., more potential particles to adhere to.

Furthermore, the surfactants used in the Grillstad tunnel are mainly non-ionic (Tab. 1), which have been found to have higher sorption affinity for particles than anionic surfactants, as used in Purify's Infravask in Smestad tunnel (Ying, 2006). Roseth and Søvik, 2006, also found tunnel wash water with high particle load to reduce *total* organic carbon and detergent concentrations more rapidly than tunnel wash water with lower particle load. Similar to the present thesis, Roseth and Søvik, 2006, could not determine if this was due to biological degradation or particle sorption. Still, the same study indicated that particle adhesion could have a more extensive effect on the total organic carbon reduction than the biological degradation.

Jurado et al., 2013 studied aerobic biodegradation of Benzenesulfonic acid (LAS), contained in Purify's Infravask, Fatty-alcohol ethoxylate (FAE) and Cocoamidopropyl betaine. The two latter groups are contained in Sanego's Solvynol detergent. The studied matrices deviate from the present thesis in that they were synthetic, allowing DOC tracing to be attributed to surfactants. The study found all three surfactants mentioned above to be 90 % biodegradable within 21 days of residence time. The results are not transferable to the present thesis, but suggest that sufficient surfactant degradation ($>$ 60 % in 28 days) could occur in the tunnel wash water matrices. This emphasises the relevance of more sophisticated biodegradation tests to be conducted on tunnel wash water than carried out in this thesis. DOC concentration reduction solely gives information about the ultimate degradation of organics to end products, whiles intermittent degradation products are also included in the DOC fraction (smaller dissolved organic carbon molecules). Alternatively, DOC concentration gives information on the adhesion of dissolved organic carbon to particles segmenting. Regardless, detergent degradation in tunnel wash water could not be documented isolated, solely supporting physicochemical characteristics of the tunnel wash water matrices (pH, DO, DOC).

### 4.4.3 pH

pH measurements were stable in all heights of the sedimentation columns for both the Grillstad tunnel (GT) and the Smestad tunnel (SmT). Top, mid and bottom taps (illustrated in Figure 4) for the same days of sedimentation differentiated 0.0 to 0.1 pH units. The mean values of top, mid, and bottom of columns containing the same samples (GT1 and GT2, SmT1 and Smt2) are therefore presented in 15. The averaged pH values of the columns gave similar, modest decrease during sedimentation from initial pH of 8.6 ± 0.01 to 7.9 ± 0.06 for column GT1 and 8.0 ± 0.07 for GT2. Thereby, pH reductions were 9 % and 8 %, respectively for the Grillstad tunnel. pH measurements of tunnel wash water from the Smestad tunnel indicated a similar pH reduction trend as observed for the Grillstad tunnel. Although, the initial pH value for the Smestad tunnel was 7.7 ± 0.02, which is lower than for the Grillstad tunnel. Sample from the Smestad tunnel then slightly increased 0.3 pH units on average until day 11 of sedimentation, before reducing again to final pH of 7.8 ± 0.06. Slightly basic initial pH values are in accordance with previous measurements in the range 7.4 to 9 (Meland and Rødland, 2018; Andersen and Vethe, 1994; Aasum, 2014; Bjotveit, 2020).

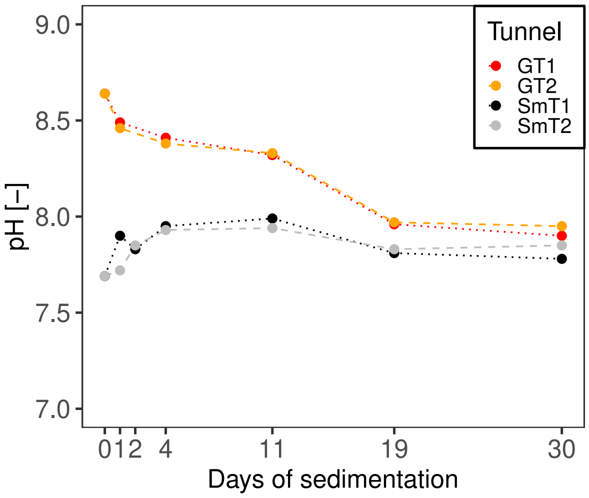


Figure 15: pH variation during sedimentation of tunnel wash water from the Grillstad and Smestad tunnels.

Slowly decreasing pH value during sedimentation can originate from biodegradation of organic material. Reduced, but consistently positive DO and DOC concentrations during sedimentation support organic material being degraded under aerobic conditions. Ultimate detergent or other organic compounds' degradation would produce $CO_2$, which can hydrate and dissociate twice in water, lowering the water's pH. Modest pH decrease during sedimentation of samples from the Grillstad and Smestad tunnels results in effluent pH values closer to what may be expected in Norwegian recipients, lowering the likelihood of environmental degradation (Miljødirektoratet, 2018).

### 4.4.4 Ion chromatography and electric conductivity

Indicators of road salts, $Na^+$ and $Cl^-$ ions, were present in considerable amounts in tunnel wash water from the Grillstad and Smestad tunnels. $Na^+$ concentrations in tunnel wash water from the Grillstad tunnel fluctuated around average concentration of 260 ± 15 mg/L throughout 30 days of sedimentation, with a modest declining trend. No clear trend was observed for $Cl^-$ concentrations. For example, the $Cl^-$ concentrations declined from initial concentration of 414 ± 2 mg/L until reaching minimum concentration of 332 ± 9 mg/L at 19 days of sedimentation, before increasing to 423 ± 14 mg/L at day 30. The ion concentrations can be observed in Figure 16.

The electric conductivity modestly increased during sedimentation of tunnel wash water from both tunnels, as plotted in Figure 16a. Several ions could affect the increase, although single ions in Figure 16 do not follow the same pattern as the electric conductivity's modest increase. Furthermore, the previously discussed increase in dissolved metal concentrations during sedimentation could impact the electric conductivity. The electric conductivity's correlations with other parameters are further discussed in Section 4.5.

The ions $F^-$, $Br^-$ and $Ca^{2+}$ were also determined in tunnel wash water samples. The $Ca^{2+}$ results were omitted here, due to standard deviations repeatedly being high, compared to determined values. This the result of identified calibration problems for the $Ca^{2+}$ curves with the ion chromatography instrument, the 940 Professional IC Vario. Despite high standard deviations, the $Ca^{2+}$ is indicated to be high in both sample from the Grillstad and the Smestad tunnel, with mean concentrations of 153 ± 53 $\mu$g/L and 175 ± 112 $\mu$g/L, respectively. The detected concentrations of $F^-$ and $Br^-$ were close to or below the detection limit of 1.0 $\mu$g/L in more than half the runs, indicating concentrations close to the detection limit in many cases. Therefore, these ions were not plotted here.

The irregularly detected ion concentrations in this section demonstrate why sedimentation as a primary treatment step is not expected to remove dissolved matter. Further study of the dissolved ions' effects of the tunnel wash water matrix is outside the scope of this thesis, but their presence in considerable concentrations (mg/L) highlights the need for further work to understand the processes occurring during sedimentation. Moreover, increased knowledge of dissolved parameters allows suitable design of secondary treatment processes.

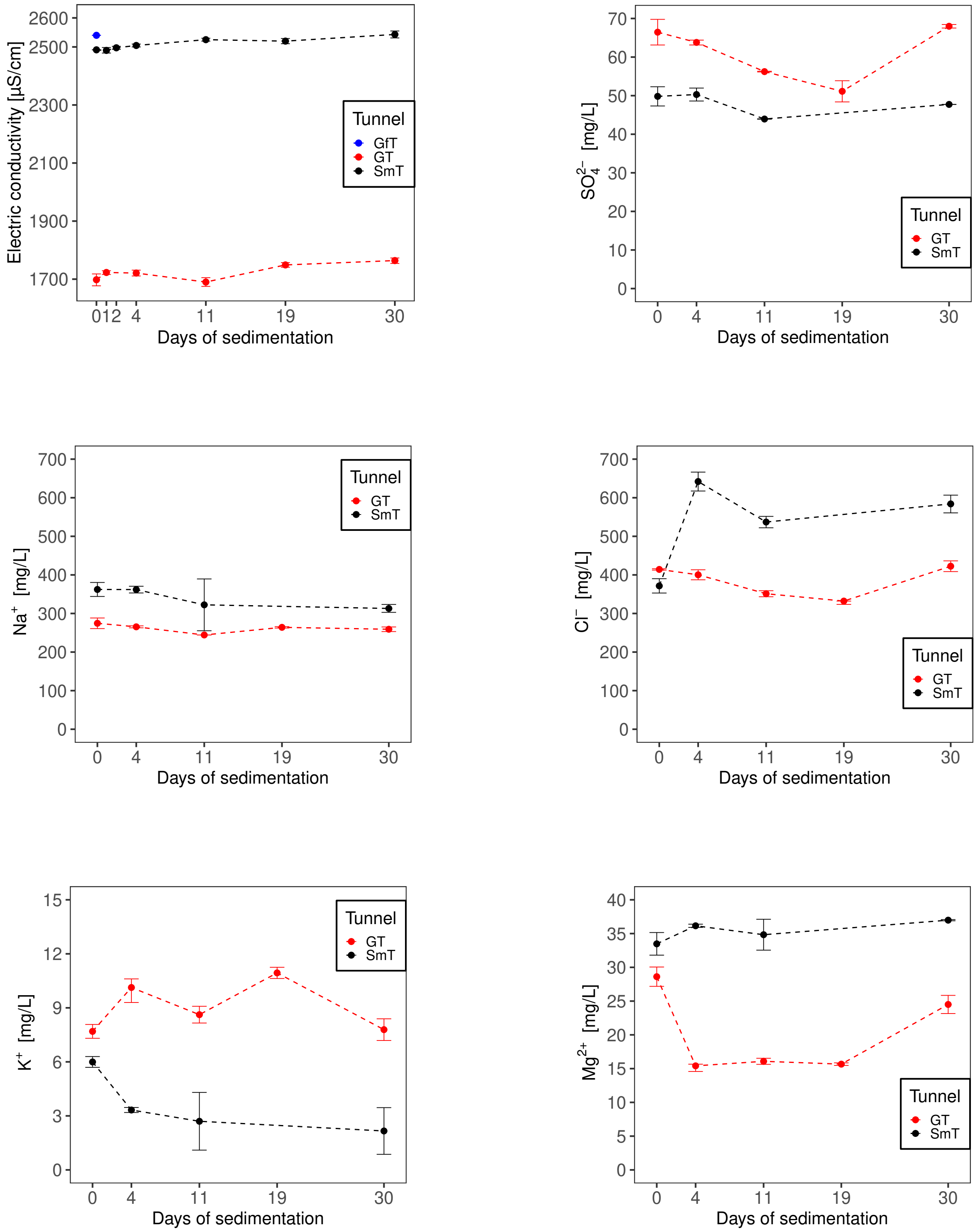


Figure 16: Electric conductivity, Sulfate (SO$^{4-}$), Sodium (Na$^+$), Chlorine (Cl$^-$), Manganese (Mg$^{2+}$) and Potassium (K$^+$) in samples from Grillstad tunnel and Smestad tunnel. Please remark the difference in the y-axes between the left and right columns.

## 4.5 Statistical analysis: Principal Component Analysis

The total metal concentrations (unfiltered fraction) were combined with the physicochemical parameters TS, TSS, turbidity, DOC, DO, pH and electric conductivity in Principal Component Analyses (PCAs). The first two principal components (axes) represented significantly more of the variability in the measurements than consecutive dimensions, with combined variance explanation of 95.8 % to 98.5 %. Therefore, the following results are represented by the variability in the first plane, allowing visualization in a single plane.

### 4.5.1 Clustering in score plots

The score plots in the top of Figure 16 visualize how initial measured values (day 0) scatter along the first principal component. After day four of sedimentation and on-wards, the spread in individuals is reduced to individuals from different days of measurement overlapping in the plane of the two first principal components. The combined score plots of the same measurements for the Grillstad and Smestad tunnels in the top of Figure 18 visualize how the individuals cluster according to tunnel. Simultaneously, the individuals from the Grillstad tunnel that disperse along the first principal component in the separate PCA in Figure 16 also disperse in the combined analysis. The PCA algorithm did not identify these measurements as outliers, but they include for example the elevated Zn concentrations detected in tunnel wash water from the Grillstad tunnel.

In the same, combined PCA, the clustering according to tunnel is mainly represented by the second principal component, which incidentally represents 27 % of the measurements' variability. The variability according to day of sedimentation (0, 4, 11, 19, 30 days) is visible along the first principal component in the second score plot in Figure 18. The initial measured water quality parameters gradually shifts towards the left, towards final measurements after 30 days, following the main declining trend of the measurements.

### 4.5.2 Loading plots

The unfiltered metals Al, As, Cu, Cr, Fe, Mn, Ni and Zn have high relative removal rate and were well represented by the first principal component (PC). The first PC in the three PCAs of unfiltered metal fractions and general water quality parameters convey more than 80% of the total variability in the measurements, as illustrated in Figure 16. The same unfiltered metal fractions were correlated to TSS; $p = 0.70$ for Zn, p-values of 0.91 to 1.0 for the remaining metals mentioned above. The metals' removal rates vary, as seen in Figure 9, but Al and Fe have similar removal rate as TSS of 99 % and consequently p-value above 0.99.

The first subsample extraction for metal detection occur after four days of sedimentation. After four days, particle load parameters TS, TSS and turbidity had already had the largest absolute reduction in particle load (Figure 6). The particle bound metal concentrations had decreased in a similar way after four days of sedimentation. Hence, this corresponding reduce is well represented theoretically by the PCA. Similar correlations between TSS and total metal concentrations were found in Meland and Rødland, 2018. The same article points out how TSS determination thereby can be utilized as costeffective indicator of the presence of heavy metal concentrations in tunnel wash water, which is also the case for the two tunnels analysed in the current thesis. Similarly, the total metal concentrations of Cd, Pb, As, Ni, Cr and Al were found to correlate well in the PCA of Aasum, 2014.

The thesis of Aasum, 2014, found Cu and Zn to correlate well with each other, but to a lesser extent with remaining metals. The same tendency can be observed in the combined PCA of the Grillstad and Smestad tunnels in Figure 18, where Cu and Zn to a less extent correlates with particle removal (turbidity, TSS) and remaining metals. This is likely the result of both Cu and Zn having variable removal efficiency (Figure 9), with total Cu concentration increasing from day 11 to day 30 in sample from the Grillstad tunnel. Meland and Rødland, 2018, however found Cu and Zn to be well correlated with remaining measurements, as they were solely measured in untreated tunnel wash water.

The PCAs performed visualize that dissolved oxygen and organic carbon (DO and DOC) and pH are weakly correlated in sample from the Grillstad tunnel in Figure 16. Opposite, the modest pH variations in the sample from the Smestad tunnel (previously seen in Figure **??**) is uncorrelated with the DO and DOC decrease, expressed as the pH vector being nearly orthogonal to DO and DOC vectors in Figure 16. These measurements for both tunnels are not well correlated in the joint PCA in Figure 18. DO and DOC are uncorrelated with the two principal components of the joint loading plot. The correlations between these three parameters are weaker than the correlations between particle and particle bound metals' removal, with highest p-value of 0.71 for DOC and pH in the combined loading plot. Furthermore, the weak correlations underlines the previous remark on the need for more sophisticated determination of organic compounds' degradation in tunnel wash water.

The PCAs suggest that pH is not correlated with most unfiltered metal fractions. However, separate PCAs were performed to include pH, ions and 1.2 $\mu$m, 0.45 $\mu$m and 3 kDa metal fractions. This was based on pH's affect on metal solubility in water, combined with the previously observed increase in several metals' dissolved fraction during sedimentation, e.g. Cu, Zn, Mn, Ni and As. The loading plots of these analyses are solely given in App. F, due to contingencies found in the correlations. The resulted in dispersed score and loading plots, where pH has no significant correlations to the 1.2 and 0.45 $\mu$m metal fractions (p-values $< 0.5$ in all cases). The exceptions were however the truly dissolved metal fractions (3 kDa) for both the Grillstad and Smestad tunnels. pH correlated with all detected truly dissolved metals, with p-values ranging from 0.82 for Ni to 0.95 for Cu. Truly dissolved As, which was below detection limits is the only exception of metals. These correlations suggest that modest pH reduction during sedimentation (as seen in Figure 15) influences the truly dissolved metal fractions' increase during sedimentation.

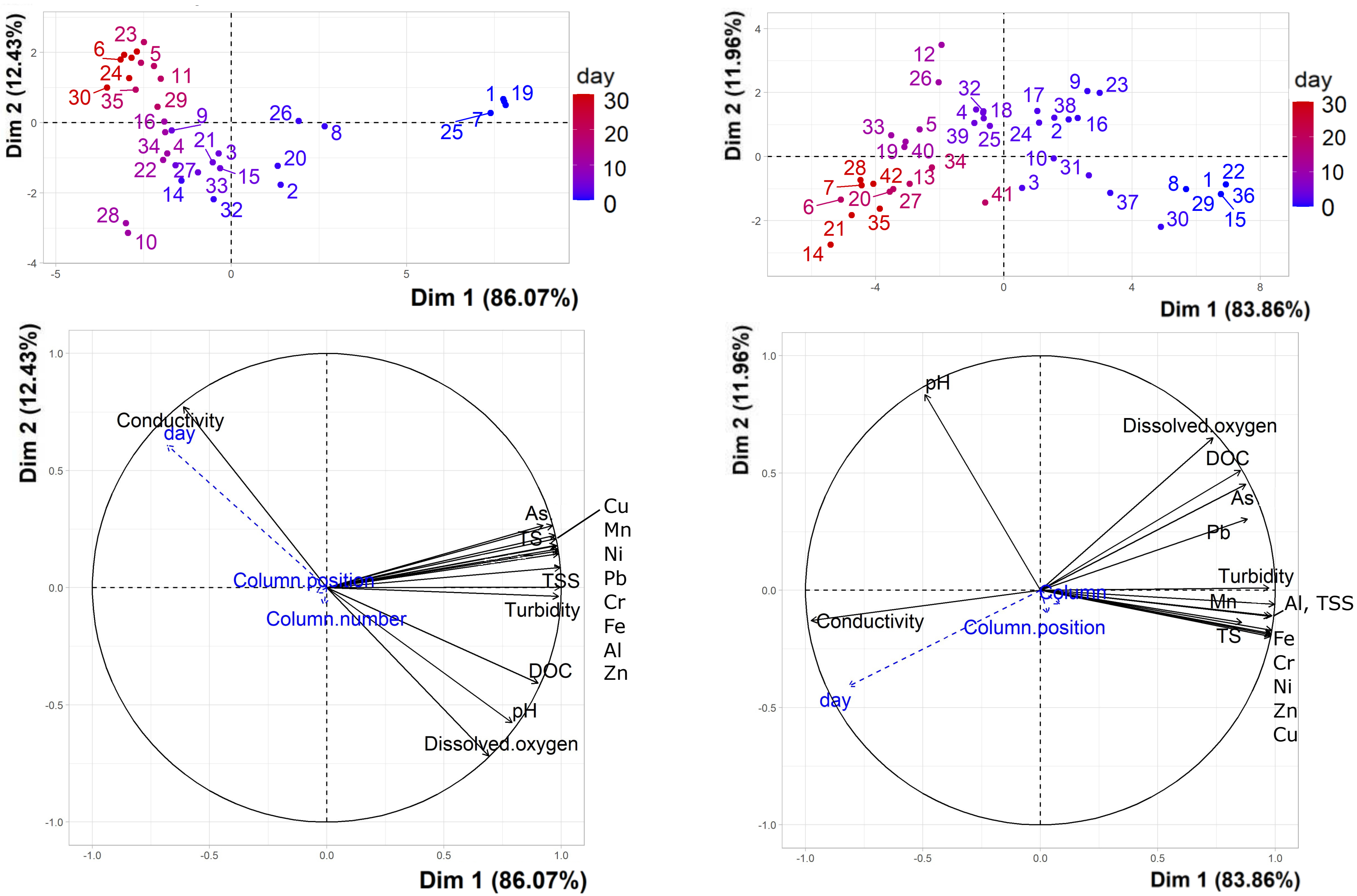


Figure 17: Score plots (top row) and loading plots (bottom row) from individual principal component analysis for the Grillstad tunnel (left) and Smestad tunnel (right). Each individual measurement is represented by a number in the score plots. Unfiltered metal concentrations at 0, 4, 11 and 30 days of sedimentation are displayed with water quality parameters turbidity, total solids (TS), and total suspended solids (TSS) pH, conductivity, dissolved oxygen and dissolved organic carbon (DOC) for the same days.

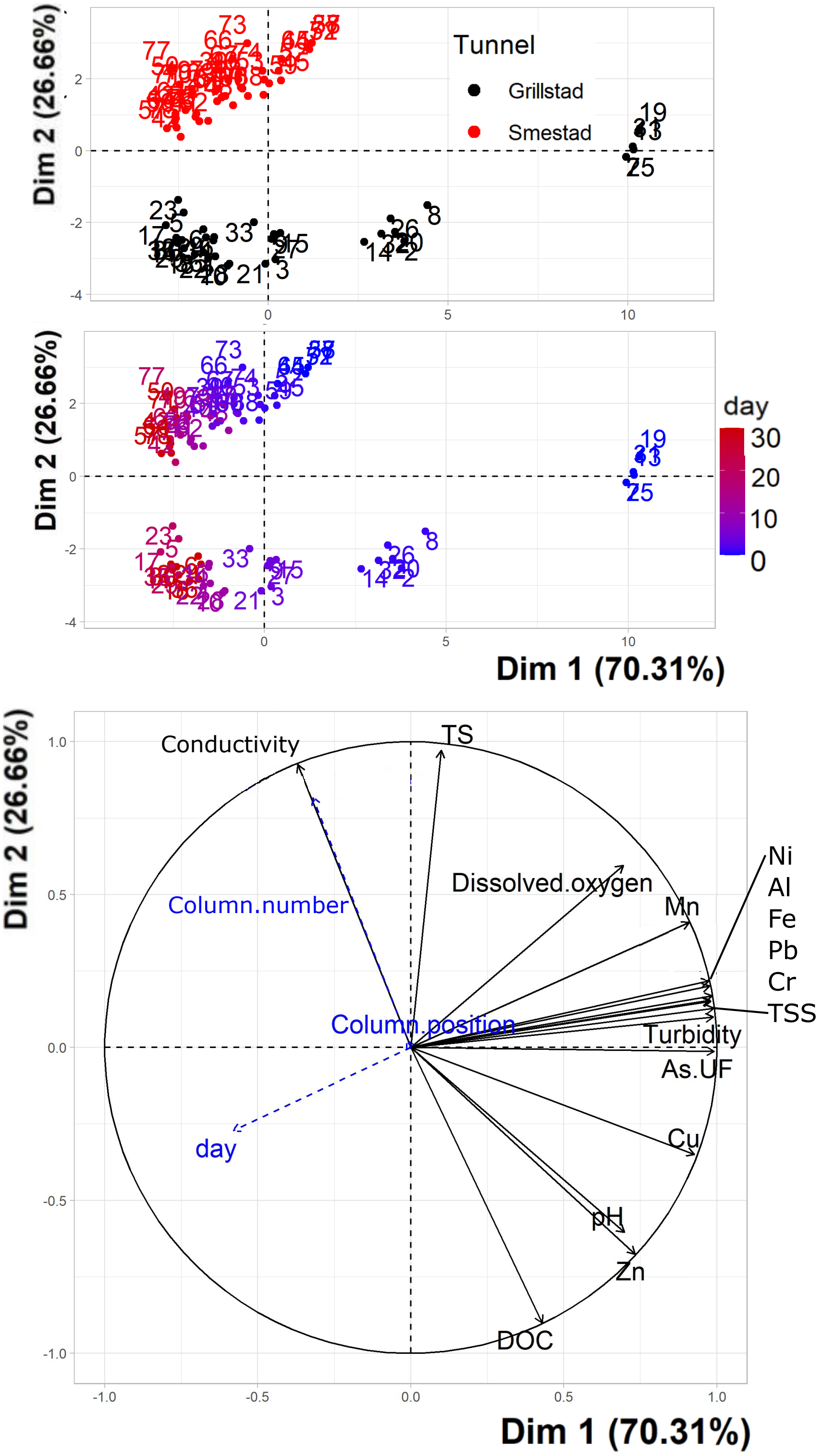


Figure 18: Top: Score plot from principal component analysis (PCA) of all water quality parameters measured in this study, illustrating their clustering. Bottom: loading plot of the same measurements. Black vectors are active (included in the main PCA). Blue vectors represent illustrative variables (excluded from the main PCA).

The last parameter of the PCAs discussed above, electric conductivity, deviate from the rest. The electric conductivity increases during sedimentation, which is why it is negatively correlated or uncorrelated with remaining parameters in the current PCAs. The electric conductivity's correlations with ions and 1.2 $\mu$m, 0.45 $\mu$m and 3 kDa metal fractions were also analyzed, to see if they better represented the variability than found for unfiltered metals and particle concentrations. The loading plots of these analyses are given in App. F, visualizing how electric conductivity is negatively or not correlated with the respective metal fractions. Lastly, the variations in electric conductivity are not well represented by the principal components found for ions concentrations and remaining water quality parameters; the vector does not reach the loading plot circumference. This contradicts the expected influence of road salts on the electric conductivity, likely explained by the fluctuating concentrations found for ions in Figure 16. In summary, the electric conductivity is found to stand out in the current measurements, with no clear correlation to other parameters.

The PCAs performed are under the assumptions made on how the water quality parameters behave during measurements. Dissolved parameters (ions, DOC, DO, electric conductivity) were assumed to be uniform over the water columns' heights, therefore measurements were extracted from the mid tap of columns. In contrast, parameters associated with particle load (turbidity, TS, TSS, PSDs) were assumed to vary over the column heights, therefore measured in top, mid (turbidity) and bottom. These assessments of particle load and dissolved parameters' distributions are based on previous experiments in the same sedimentation pilot with samples from the Tåsen tunnel (Oslo), the Strindheim tunnel (Trondheim) and the thesis of (Bjotveit, 2020). Dissolved parameters were stable in top, mid and bottom of the sedimentation columns. Although the samples from the Tåsen and Strindheim tunnels were later found to be of unknown origin (to be discussed in Section 4.6). Consequently, the input to the PCA algorithm is already characterized by subjective assumptions prior to analysis of their correlations.

## 4.6 Operation and management of sampled sedimentation basins

Unintended findings from sampled sedimentation basins' are summarized in this section, to highlight how practical difficulties during the operation phase of sedimentation basins may cause reduced treatment efficiency. At worst, direct discharge of polluted tunnel wash water could be the consequence.

Sufficient operation and management of sedimentation basins is a prerequisite for satisfactory treatment efficiency for tunnel wash water. The attempted samplings in this study yielded three verified samplings, one from each of the Grillstad, Smestad and Granfoss tunnels. The rest of samplings were not possible to verify or were confirmed not to be tunnel wash water; four samplings in the Tåsen tunnel (TT1, TT2, TT3, TT4) and one in the Strindheim tunnel (ST). The considerable differences in samples' water quality are summarized by turbidity and metal concentration variations in Figure 19.

The Bjørnegård tunnel research and development plant was intended sampled two nights in a row. The first night, system failure led to insufficient tunnel wash water collection in the sedimentation basin. Samples taken were therefore discarded. The tunnel wash water did not reach the sedimentation basin the following night, meaning it was led else-were in the system. The system failure was not further investigated.

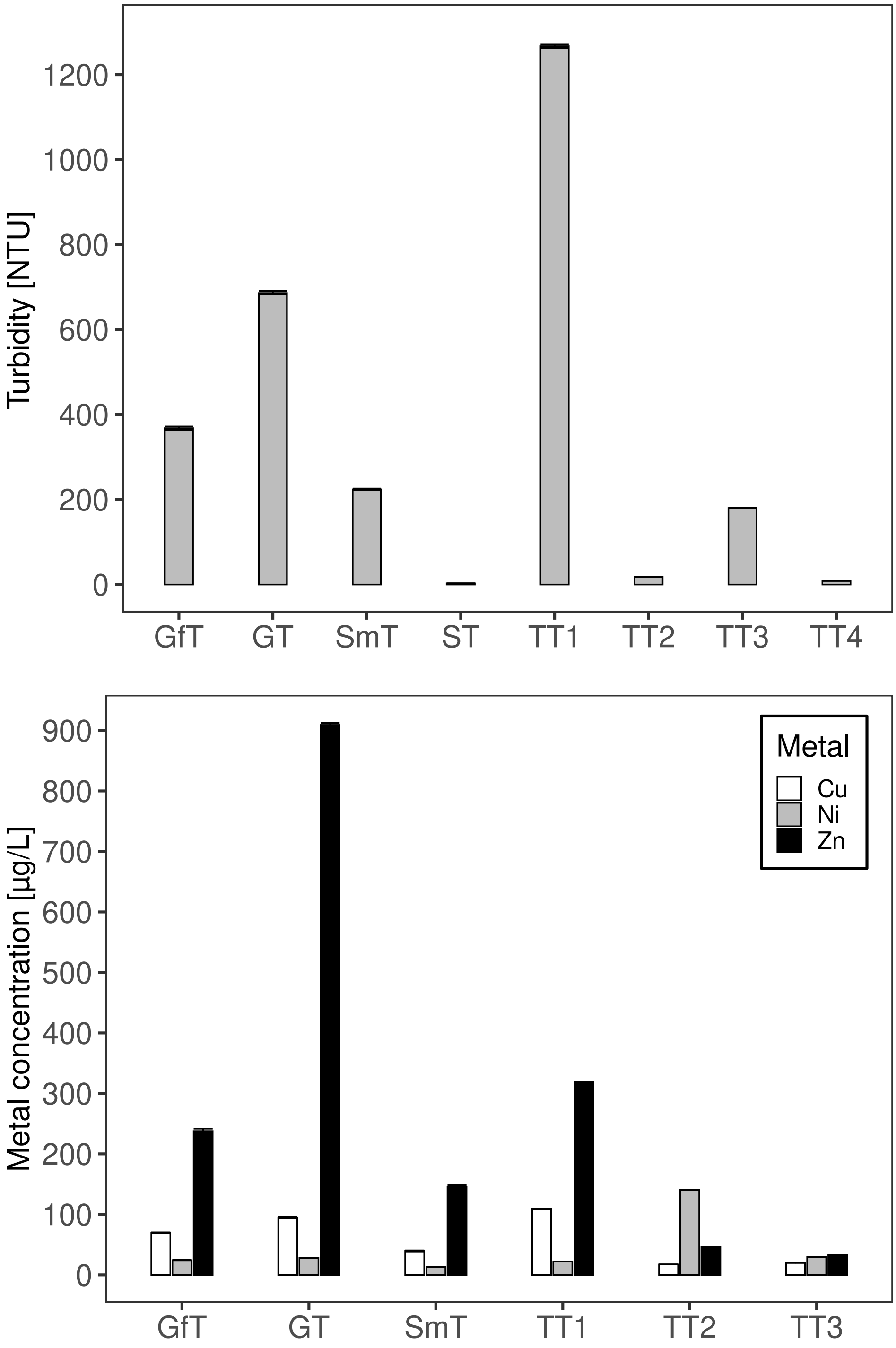


Figure 19: Turbidity and metal concentrations in samplings from the Granfoss tunnel (GfT), Grillstad tunnel (GT), the Smestad tunnel (SmT) and three sampling from the Tåsen tunnel (TT1 - TT3). The three latter have uncertainty attached to what water that was sampled.

Samplings of water from the Tåsen tunnel's sedimentation basins diverged significantly in quality. The sedimentation basin collecting tunnel wash water from the westbound tunnel run mainly contained sludge at samplings (TT1: 20.01.21, TT3: 15.03.21), diluted by an unknown amount of drainage water from the tunnel. The sludge from the first sampling (TT1) then sedimented approximately two months until the second sampling TT3, as illustrated by turbidity and metal concentrations' decrease in Figure 19. In contrast, the sedimentation basin for the eastbound direction contained water far less polluted than expected ranges for tunnel wash water (TT2: 08.02.21, TT4: 15.03.21.). The turbidity of these samplings were significantly lower than the turbidity of samplings TT1 and TT3.

The significant variations in tunnel wash water quality from the Tåsen tunnel were briefly checked. The functionality of the treatment facility during three to four months was then found to be lower than desirable. The monitoring was further hampered by technical problems with the automatic operation software. In consequence, extraneous water was contained for unknown retention time in the sedimentation basin of the Tåsen tunnel eastbound run. The first sampling from this basin, TT2, was thereby most likely extraneous water. The second sampling from the same basin was supposedly tunnel wash water that had sedimented nine days prior to sampling, due to shifts in washing plans. This is however contradicted by numerous living water organisms observed in the samples from the same sedimentation basin of the eastbound tunnel run (TT2, TT4), given in App. A, Figure 22. This is unexpected in an indoor concrete basin. The organisms' presence is more questionable due to elevated concentrations of several heavy metals in the samples they lived in, overshooting MAC-QS thresholds significantly. This is exemplified by Cu, Ni and Zn in Figure 19. The organisms were not further examined, as that is outside the scope of this thesis.

Similar to the sedimentation basin for the eastbound tunnel run of the Tåsen tunnel, sampling from the Strindheim tunnel (ST), Trondheim, was questionable. The sedimentation basin had significantly higher water table than the rest, where the basin floor was visible through 5 m water depth (25.03.21). The sample contained negligible amounts of particles, expressed as turbidity of 2.7 NTU in Figure 19. The conditions were found to be unchanged when the sedimentation basin was revisited 13.05.21. The hypothesis then raises if the sampled basin is for the water draining from the rock lining of the tunnel, as it contains elevated concentrations of sulfate ($SO_4$), nitrate ($NO_3$), potassium ($K^+$) and to some extent calcium ($Ca^+$) compared to the rest of samplings. Road salt indicating ions' concentrations, $Na^+$ and $Cl^-$, were also lower than the rest of tunnels studied. In next order, if the unidentified water in the sedimentation basin of the Strindheim tunnel would not be tunnel wash water, where is the tunnel wash water then drained or discharged.

The encountered difficulties in asset management of sedimentation basins for tunnel wash water seems to follow previously studied performance of similar assets. Åstebøl et al., 2013, studied the treatment ability of *road runoff* in 26 wet sedimentation ponds in Norway. Although not directly transferable to basins treating tunnel wash water, the study found that the majority of these sedimentation ponds have reduced functionality. Solely five out of the 26 studied ponds delivered what was defined as "good" treatment efficiency. This was due to deviations from design in executed construction and insufficient operation and maintenance. The unintended findings of the present thesis indicates that unsatisfactory asset management may be considerably present for sedimentation basins for tunnel wash water as well.

The plotted metal concentrations in Figure 19 exemplify the variations in samples, which are all significantly higher than the MAC-QS thresholds. The significant Zn concentrations in the sample from the Grillstad tunnel is overshooting sludge (TT1), highlighting why it poorly correlated to other metals in the PCA discussed previously. The high and varying metal concentrations underlines why it is crucial to control the treatment facilities, so that they can deliver their function of reducing particle load and particulate bound metals concentrations. Furthermore, to control where the tunnel wash water is discharged, avoiding direct discharge and thereby loosing the treatment effect of installed sedimentation basins. Further work on asset management of sedimentation basins is therefore recommended, to increase the environmental profit from the investments in treatment facilities. In addition, to ensure that the obligations under EU's Water Framework Directive are met, by lowering the risk of untreated discharge or lowered treatment efficiency.

# 5 Conclusion

Tunnel wash water is a complex matrix where the components' interactions are not yet fully understood. Therefore, this thesis contributed by documenting particle load reduction, heavy metal fraction concentrations, and dissolved ions during sedimentation. The results show that the particle load reduced efficiently, expressed as removal rates ranging from 94 to 99 % for turbidity, total suspended solids (TSS; $> 0.45$ $\mu$m), and the total number of particles. The particle size distributions show that the particles smaller than 1 $\mu$m dominate in numbers for the studied tunnels, which also sedimented to a small extent. The latter is expressed as modest reduction of the total solids (TS; all particle diameters) ranging from 17 to 35 %. This was mainly due to the TSS reduction. Further classification of the particle removal is hindered by the lack of corresponding recipient thresholds. The sedimented tunnel wash water quality and particle load were stable and homogeneous over 2 m water columns, suggesting that sedimentable particles are removed within 30 days, but a further reduction of the bulk of particles ($< 1.0$ $\mu$m) needs a second treatment step.

The total metal concentrations also declined, ranging from 49% reduction of Zn to 99.7% Al reduction. These reductions were well correlated with TSS, with p - values of 0.91 to 1.0 for seven of ten metals (Al, As, Cu, Cr, Fe, Mn, Ni). The colloidal (1.2 $\mu$m to 3 kDa) and truly dissolved ($< 3$ kDa) metal fractions were persistent (Al, Cr, Fe, Pb) or modestly increasing (Cd, Cu, Mn, Ni, Zn) during sedimentation. Simultaneously, pH reduced to slight basic effluent values of 7.8 to 8.0, correlated to the persistent and increasing truly dissolved fractions of all measured metals (p - values range from 0.82 to 0.95). After 30 days of sedimentation, the concentrations of Cu, Zn, As, and Cr are higher than the environmental quality standards for recipient short time exposure. Therefore, the recipients' dilution effects and vulnerability should be studied to classify the potential environmental harm, especially since the bulk of effluent Cu, Zn, and As concentrations are in the bio-available, dissolved form, increasing the risk of heavy metal bio-accumulation.

Indicators of biological degradation, dissolved oxygen (DO) and dissolved organic carbon (DOC), were reduced during sedimentation. DO concentration declined 37 to 60 % , to sedimented quality ranging from 4.78 to 7.56 mg/L. Aerobic conditions were present in the whole water columns. However, anaerobic conditions are present to a larger extent in the sedimentation basins in situ. The DOC modestly reduced 14 to 20 % to mean concentrations of 10.2 to 25.4 mg/L, of which an undetermined fraction may be detergents or intermittent degradation products. The reduction during sedimentation can be attributed to particle adhesion, aerobic degradation, or a combination. This implicates the need for detergent determination to follow the degradation process.

Specifications of what water quality primary and secondary treatment should yield are a prerequisite to decide whether secondary treatment is necessary to fulfill the obligations of the Vannforskriften, 2007. Lastly, sedimentation basins depend on proper asset management to achieve pollutant reductions, but currently this was inadequate in three out of six studied treatment facilities. Most crucial is the uncertainty in where the tunnel wash water is discharged in individual cases. This highlights that sedimentation as a primary treatment step is not always successful in practice.

## 5.1 Recommendations for further work

- Sample sedimented tunnel wash water, to check how the custom made sedimentation pilot used in this study corresponds to in situ tunnel wash water quality. The attempted samplings of this study were not representative.
- Document to what extent biological degradation occur in tunnel wash water, i.e., by checking if bacteria growth (concentrations) are opposite proportional to dissolved carbon and oxygen (DOC and DO) concentrations during sedimentation. Flow cytometry analysis or analogous methods could be further investigated for the purpose.
- In depth study of a suitable secondary treatment step than removes remaining dissolved matter and particles smaller than 1 $\mu$m. Here, it must be remarked that this work is initiated in the on-going research project the current thesis is a part of.
- Study and develop the metal fractionation further, to determine the causes of detected metal leaching after filtration. Firstly, by including minimum three blank samples of all filter sizes. Possible alternatives to the used filtration could be cross flow filtering or stirred Amicon® cells.
- Develop further recipient limit values of water quality parameters, expanding it from currently being mainly heavy metals and polycyclic aromatic hydrocarbons (PAHs). This could allow to better evaluate the environmental effect of tunnel wash water discharge and better design of primary and secondary treatment steps.
- Study how asset management affect treatment efficiency of sedimentation basins. That is, if the differences unintentionally found in this thesis are representative. This could strengthen the basis for making adequate operational and maintenance plans, which promotes reliable treatment throughout the life span of the assets.

In depth study of detergent, PAHs and other organic substances' degradation are also desirable for further work. The topics are currently studied in an associated doctoral thesis (H. Vistnes, NTNU, expected published in 2024). More sophisticated analyses aim to connect parameters measured in this thesis, which solely indicate detergent degradation (dissolved oxygen and carbon, pH) with detergent concentrations and intermittent degradation products.

# 6 References

Aasum, Jon-Henning. 2014. *Effekter av vaskemiddel (TK601) på mobilitet av metaller ved sedimentering av tunnelvaskevann fra Nordbytunnelen (E6), Ås kommune, Akershus : et laboratorieforsøk.* Ås, Norway: Norwegian University of Life Sciences, November. Accessed February 5, 2021. https://nmbu.brage.unit.no/nmbu-xmlui/handle/11250/189649.

*About NORWAT* [in no]. 2016, June. Accessed May 7, 2021. https://www.vegvesen.no/en/professional/focus-areas/research-and-development/completed-projects/norwat.

Adamiec, Ewa, Elbieta Jarosz-krzemiska, and Robert Wieszaa. 2016. "Heavy metals from non-exhaust vehicle emissions in urban and motorway road dusts" [in English]. *Environmental Monitoring and Assessment* 188, no. 6 (June): 1–11. Accessed April 21, 2021. https://doi.org/http://dx.doi.org/10.1007/s10661-016-5377-1.

Allan, Ian J., Steven G. O'Connell, Sondre Meland, Kine Bæk, Merete Grung, Kim A. Anderson, and Sissel B. Ranneklev. 2016. "PAH Accessibility in Particulate Matter from Road-Impacted Environments." Publisher: American Chemical Society, *Environmental Science & Technology* 50, no. 15 (August): 7964–7972. Accessed May 6, 2021. https://doi.org/10.1021/acs.est.6b00504.

Andersen, S., and Ø. Vethe. 1994. "Mobilisation of heavy metals during tunnel maintenance" [in en]. *Science of The Total Environment,* Highway Pollution, 146-147 (June): 479–483. ISSN: 0048-9697, accessed November 19, 2020. https://doi.org/10.1016/0048-9697(94)90272-0.

Åstebøl, S. O., K. A. Paus, S. Robbe, V. Ulland, and E. Lausund. 2013. *Tilstanden til rensebassenger i Norge (The condition of retention ponds in Norway).* Report. Accepted: 2020-06-12. Statens vegvesen, November. Accessed October 15, 2020. https://vegvesen.brage.unit.no/vegvesen-xmlui/handle/11250/2657795.

Barra Caracciolo, Anna, Martina Cardoni, Tanita Pescatore, and Luisa Patrolecco. 2017. "Characteristics and environmental fate of the anionic surfactant sodium lauryl ether sulphate (SLES) used as the main component in foaming agents for mechanized tunnelling." *Environmental Pollution* 226 (July): 94–103. Accessed February 4, 2021. https://doi.org/10.1016/j.envpol.2017.04.008.

Benson, Bruce B., and Daniel Krause. 1980. "The Concentration and Isotopic Fractionation of Gases Dissolved in Freshwater in Equilibrium with the Atmosphere. 1. Oxygen." *American Society of Limnology and Oceanography* 25 (4): 662–671. ISSN: 0024-3590, accessed May 1, 2021. https://www.jstor.org/stable/2835754.

Bjordal, Silje. 2020. "Master's thesis, Silje Bordal." Master's thesis, NTNU, February. Accessed October 16, 2020.

Bjotveit, Ågot. 2020. "Investigation of Particle and Heavy Metal Removal and Detergent Degradation in Sedimented Tunnel Wash Water." Master's thesis, NTNU, June. Accessed September 23, 2020.

Bonilla-Petriciolet, A., and D. I. Mendoza-Castillo. 2017. *Adsorption Processes for Water Treatment and Purification.* Mexico: Springer International Publishing AG. ISBN: 978-3-319-58135-4.

*Coulter LS Series Product Manual.* 2011. Accessed October 17, 2020. https://www.beckmancoulter.com/wsrportal/techdocs?docname=4237214EA.pdf.

Dyer, Scott D., Hans Sanderson, Scott W. Waite, Remi Van Compernolle, Bradford Price, Allen M. Nielsen, Alex Evans, Alvaro J. Decarvalho, Dennis J. Hooton, and Andrew J. Sherren. 2006. "ASsessment Of Alcohol Ethoxylate Surfactants And Fatty Alcohols Mixtures In River Sediments And Prospective Risk Assessment." *Environmental Monitoring and Assessment* 120, no. 1 (September): 45–63. Accessed January 29, 2021. https://doi.org/10.1007/s10661-005-9048-x.

Firestone, Jordan A., and Sidney M. Gospe. 2009. "Clinical Neurotoxicology, Chap. 35 - Organic Solvents" [in en]. In *Clinical Neurotoxicology,* edited by MICHAEL R. Dobbs, 401–414. Philadelphia: W.B. Saunders, January. ISBN: 978-0-323-05260-3, accessed May 4, 2021. https://www.sciencedirect.com/science/article/pii/B9780323052603500411.

Frost, K., and C. Gremmertsen. 2019. *Forvaltningsplan med driftsinstruks for Smestadtunnelen renseløsning,* May. Accessed January 14, 2021.

Garshol, F. 2016. *Laboratorietester - rensing av vaskevann fra Nordbytunnelen* [in nob]. Report 521. Tex.ids= garshol_laboratorietester_2016 tex.copyright: Attribution-NoDerivatives 4.0 Internasjonal. Norway: Norwegian Public Road Administration, March. Accessed October 6, 2020. https://vegvesen.brage.unit.no/vegvesen-xmlui/handle/11250/2671290.

Gremmertsen, C., and K. Frost. 2015. "Håndtering av tunnelvaskevann i eksisterende tunneler – Eksempel fra rehabiliteringen av Smestadtunnelen." *Vannforeningen,* 5. Accessed January 14, 2021. https://vannforeningen.no/wp-content/uploads/2015/06/2015_924565.pdf.

Grung, M., K. Petersen, E. Fjeld, I. Allan, J. H. Christensen, L. Malmqvist, S. Meland, and S. Ranneklev. 2016. "PAH related effects on fish in sedimentation ponds for road runoff and potential transfer of PAHs from sediment to biota" [in en]. *Science of The Total Environment* 566-567 (October): 1309–1317. ISSN: 0048-9697, accessed November 18, 2020. https://doi.org/10.1016/j.scitotenv.2016.05.191. http://www.sciencedirect.com/science/article/pii/S0048969716311299.

Hallberg, M., G. Renman, L. Byman, G. Svenstam, and M. Norling. 2014. "Treatment of tunnel wash water and implications for its disposal." *Water Science and Technology* 69, no. 10 (March): 2029–2035. ISSN: 0273-1223, accessed May 6, 2021. https://doi.org/10.2166/wst.2014.113. https://doi.org/10.2166/wst.2014.113.

*Håndbok R610 Standard for drift og vedlikehold av riksveger.* 2014. The Handbook series of the Norwegian Public Roads Administration. Oslo, Norway: Norwegian Public Roads Administration. ISBN: 978-82-7207-655-8. https://www.vegvesen.no/_attachment/61430/binary/964067.

ISO. 2017. *General requirements for the competence of testing and calibration laboratories (ISO/IEC 17025:2017).* 3.0. International Organization for Standardization, November. https://www.standard.no/no/Nettbutikk/produktkatalogen/Produktpresentasjon/?ProductID=958400.

*ISO 10523.* 2008. International Standard. Switzerland: Water quality — Determination of pH. Second edition. Accessed October 17, 2020. https://www.standard.no/no/Nettbutikk/produktkatalogen/Produktpresentasjon/?ProductID=529464.

*ISO 5814.* 2012. European Committee for Standardization. Water quality — Determination of dissolved oxygen — Electrochemical probe method. Accessed February 8, 2020. https://www.standard.no/no/Nettbutikk/produktkatalogen/Produktpresentasjon/?ProductID=606913.

*ISO 7027.* 2016. Water quality Determination of turbidity Part1: Quantitative methods (I. Accessed January 17, 2021. https://www.standard.no/no/Nettbutikk/produktkatalogen/Produktpresentasjon/?ProductID=832960.

*ISO 8245:1999.* 1999. International Organization for Standardization (ISO): Water quality — Guidelines for the determination of total organic carbon (TOC) / dissolved organic carbon (DOC). Accessed January 17, 2021. https://www.standard.no/no/Nettbutikk/produktkatalogen/Produktpresentasjon/?ProductID=124062.

Johansen, Susanne Lund, and Helene Thygesen. 2013. *Økologisk risikovurdering med biotisk ligandmodell : En feltstudie av tunnelvaskevann og vegavrenning i rensebasseng* [in NOB]. Report 230. Norway: Norwegian Public Road Administration, July. Accessed September 19, 2020. https://hdl.handle.net/11250/2657883.

Jurado, Encarnación, Mercedes Fernández-Serrano, and Francisco Ríos and Manuela Lechuga. 2013. *Aerobic Biodegradation of Surfactants* [in en]. Vol. Chapter 3: Open access peer-reviewed. Publication Title: Biodegradation - Life of Science. IntechOpen, June. ISBN: 978-953-51-1154-2, accessed May 10, 2021. https://doi.org/10.5772/56120. https://www.intechopen.com/books/biodegradation-life-of-science/aerobic-biodegradation-of-surfactants.

Klöckner, Philipp, Bettina Seiwert, Steffen Weyrauch, Beate I. Escher, Thorsten Reemtsma, and Stephan Wagner. 2021. "Comprehensive characterization of tire and road wear particles in highway tunnel road dust by use of size and density fractionation" [in en]. *Chemosphere* 279 (September): 130530. ISSN: 0045-6535, accessed June 14, 2021. https://doi.org/10.1016/j.chemosphere.2021.130530. https://www.sciencedirect.com/science/article/pii/S0045653521010018.

Kowollik, H. S. 2020. "A study of size fractions of metals in sedimented tunnel wash water." Master's thesis, NTNU, June. Accessed January 28, 2020.

Meland, S. 2010. "Ecotoxicological effects of highway and tunnel wash water runoff." Accepted: 2013-12-13T08:39:07Z Publication Title: 86 s. PhD diss., [Doctoral thesis (PhD)]. Norwegian University of Life Sciences. Accessed December 2, 2020. https://vegvesen.brage.unit.no/vegvesen-xmlui/handle/11250/190880.

Meland, S. 2012. "Tunnelvaskevann – En kilde til vannforurensning." *Vannforeningen,* no. 02, 182–193. https://vannforeningen.no/wp-content/uploads/2015/06/2012_853040.pdf.

Meland, S., and E. S. Rødland. 2018. "Forurensning i tunnelvaskevann - en studie av 34 veitunneler i Norge." Tex.ids= meland_forurensning_2018, *Vann* 53, no. 1 (January): 54–65. Accessed September 16, 2020. http://hdl.handle.net/11250/2597273.

Meland, Sondre, Reidar Borgstrøm, Lene Sørlie Heier, Bjørn Olav Rosseland, Oddvar Lindholm, and Brit Salbu. 2010. "Chemical and ecological effects of contaminated tunnel wash water runoff to a small Norwegian stream" [in en]. *Science of The Total Environment* 408 (19): 4107–4117. Accessed November 7, 2020. https://doi.org/10.1016/j.scitotenv.2010.05.034.

Midtbø, Mia Kristin. 2016. *Slik ser tunnelen ut før og etter vask* [in nb], March. Accessed December 19, 2020. https://www.adressa.no/nyheter/trondheim/2016/03/11/Slik-ser-tunnelen-ut-f%C3%B8r-og-etter-vask-12255111.ece.

Miljødirektoratet. 2018. *Veileder 02:2018 Klassifisering av miljøtilstand i vann. Økologisk og kjemisk klassifiseringssystem for kystvann, grunnvann, innsjøer og elver.* [in NO], January. Accessed November 6, 2020. https://www.vannportalen.no/veiledere/klassifiseringsveileder/.

Miljødirektoratet, COWI Aquateam, NIVA, and NGI. 2020. *Grenseverdier for klassifisering av vann, sediment og biota – revidert 30.10.2020* [in NOB], October. Accessed January 28, 2020. https://www.miljodirektoratet.no/publikasjoner/2016/september-2016/grenseverdier-for-klassifisering-av-vann-sediment-og-biota/.

Mketo, Nomvano, Philiswa N. Nomngongo, and J. Catherine Ngila. 2016. "An innovative microwave-assisted digestion method with diluted hydrogen peroxide for rapid extraction of trace elements in coal samples followed by inductively coupled plasma-mass spectrometry" [in en]. *Microchemical Journal* 124 (January): 201–208. ISSN: 0026-265X, accessed March 23, 2021. https://doi.org/10.1016/j.microc.2015.08.010. https://www.sciencedirect.com/science/article/pii/S0026265X15001824.

Mouni, Lotfi, Djoudi Merabet, Didier Robert, and Abdelkrim Bouzaza. 2009. "Batch studies for the investigation of the sorption of the heavy metals Pb2+ and Zn2+ onto Amizour soil (Algeria)" [in en]. *Geoderma* 154, no. 1 (December): 30–35. Accessed January 26, 2021. https://doi.org/10.1016/j.geoderma.2009.09.007.

*NS 4733* [in SV]. 1983. Water analysis - Determination of suspended solids in waste water / their residue on ignition. Standard Norge. Accessed October 18, 2020. https://www.standard.no/no/Nettbutikk/produktkatalogen/Produktpresentasjon/?ProductID=134366.

*NS-EN ISO 10304-1.* 2009. Standard Norge: Bestemmelse av oppløste anioner med væskekromotografi Del 1: Bestemmelse av bromid, klorid, fluorid, nitrat, nitritt, fosfat og sulfat. https://www.standard.no/no/Nettbutikk/produktkatalogen/Produktpresentasjon/?ProductID=383003.

*NS-EN ISO 14911.* 1998. Standard Norge: Bestemmelse av oppløst Li+, Na+, NH4+, K+, Mn2+, Ca2+, Mg2+, Sr2+ og Ba2+ ved ionekromotografi Metode for vann og avløpsvann. https://www.standard.no/no/Nettbutikk/produktkatalogen/Produktpresentasjon/?ProductID=144019.

*NS-EN ISO 17294-2.* 2016. CEN: European Committee for Standardization: Water quality - Application of inductively coupled plasma mass spectrometry (ICP-MS) - Part 2: Determination of selected elements including uranium isotopes (ISO 17294-2:2016). https://www.standard.no/no/Nettbutikk/produktkatalogen/Produktpresentasjon/?ProductID=857613.

*NS-ISO 7888* [in NO]. 1985. Water quality Determination of electrical conductivity. Standard Norge. Accessed October 17, 2020. https://www.standard.no/no/Nettbutikk/produktkatalogen/Produktpresentasjon/?ProductID=146721.

*NS4764* [in NO]. 1980. Water analysis. Total residue, / total fixed in water, sludge / sediments. Standard Norge. Accessed October 18, 2020. https://www.standard.no/no/Nettbutikk/produktkatalogen/Produktpresentasjon/?ProductID=134402.

Ødegaard, Hallvard. 2013. *Vann - og avløpsteknikk* [in NO]. 2nd ed. Trondheim, Norway: Norsk Vann. ISBN: 978-82-414-0356-9.

*Operating manual Oxi 3310 IDS.* 2014. Accessed May 3, 2021. http://www.geotechenv.com/Manuals/WTW_Manuals/WTW_Oxi_3310_IDS.pdf.

*Particle Measurements - NI Vision 2015 Concepts Help - National Instruments, n.d.* Accessed April 28, 2021. https://zone.ni.com/reference/en-XX/help/372916T-01/nivisionconcepts/particle_measurements/.

*pn3000-specifications - Postnova Analytics, n.d.* [in en]. Accessed April 23, 2021. https://www.postnova.com/pn3000-specifications.html.

Rapp, Bastian E. 2017. "Microfluidics: Modelling, Mechanics and Mathematics" [in en]. In *Microfluidics: Modelling, Mechanics and Mathematics,* 421–444. Micro and Nano Technologies. Oxford: Elsevier, January. ISBN: 978-1-4557-3141-1, accessed April 22, 2021. https://doi.org/10.1016/B978-1-4557-3141-1.50020-4. https://www.sciencedirect.com/science/article/pii/B9781455731411500204.

Roseth, R., and A.K. Søvik. 2006. "Nedbryting av såper til tunnelvask." *Statens Vegvesen, Utbyggingsavdelingen,* no. UTB rapport 2006/01, accessed January 19, 2021. https://www.vegvesen.no/_attachment/203835/binary/394638?fast_title=Nedbryting+av+s%C3%A5per+til+tunnelvask.pdf.

Roseth, Roger. 2013. "Ny E6 Minnesund–Espa. Utslipp av renset vaskevann fra vegtunneler. Vurdering av resipienteffekter." P. 9-40 in PDF linked in URL, *Bioforsk Rapport* 8, no. 105 (October): 30. Accessed February 8, 2021. https://www.statsforvalteren.no/contentassets/fca71d3b482c4fba82d087148956d629/statens-vegvesen-soknad-utslipp-espatunnelen.pdf.

Selbig, Willliam R., Michael N. Fienen, Judy A. Horwatich, and Roger T. Bannerman. 2016. "The Effect of Particle Size Distribution on the Design of Urban Stormwater Control Measures" [in en]. Number: 1 Publisher: Multidisciplinary Digital Publishing Institute, *Water* 8, no. 1 (January): 17. Accessed June 8, 2021. https://doi.org/10.3390/w8010017. https://www.mdpi.com/2073-4441/8/1/17.

Strømberg, Merethe Arntsen. 2020. "Sediment removal performance of a hydrodynamic vortex separator" [in no]. Master's thesis, NTNU. Accessed October 17, 2020.

Torp, M., and S. Meland. 2013. *Estimering av forurensning i tunnel og tunnelvaskevann* [in nob]. Report. Accepted: 2018-08-09T13:49:07Z ISSN: 1893-1162 Journal Abbreviation: Estimation of pollution in wash water from tunnels Publication Title: 36. Statens vegvesen, November. Accessed December 3, 2020. https://vegvesen.brage.unit.no/vegvesen-xmlui/handle/11250/2508287.

Vannforskriften. 2007. *Forskrift om rammer for vannforvaltningen (FOR-2006-12-15-1446), Vedlegg VII E* [in NO]. Accessed June 5, 2021. https://lovdata.no/forskrift/2006-12-15-1446.

Vegdirektoratet. 2018. *Håndbok N200 Vegbygging (juli 2018)* [in NO]. July. ISBN: 978-82-7207-723-4, accessed November 2, 2020. https://www.vegvesen.no/_attachment/2364236/binary/1269980?fast_title=H%C3%A5ndbok+N200+Vegbygging+%2810+MB%29.pdf.

Wallace, Brian, Mike Purcell, and Joe Furlong. 2002. "Total organic carbon analysis as a precursor to disinfection byproducts in potable water: Oxidation technique considerations." *Journal of environmental monitoring : JEM* 4 (January): 35–42. https://doi.org/10.1039/B106049J.

Ying, G. 2006. "Fate, behavior and effects of surfactants and their degradation products in the environment." *Environment International* 32, no. 3 (January): 417–431. Accessed January 29, 2021. https://doi.org/10.1016/j.envint.2005.07.004.

Yun, Yeobog, Haemi Park, Leehyung Kim, and Seokoh Ko. 2010. "Size distributions and settling velocities of suspended particles from road and highway" [in en]. *KSCE Journal of Civil Engineering* 14, no. 4 (July): 481–488. ISSN: 1226-7988, 1976-3808, accessed June 8, 2021. https://doi.org/10.1007/s12205-010-0481-1. http://link.springer.com/10.1007/s12205-010-0481-1.

Zhang, Siyu, and Mingxuan Zhou. 2020. "Comparison of DMA-80 and ICP-MS Combined with Closed-Vessel Microwave Digestion for the Determination of Mercury in Coal" [in en]. Publisher: Hindawi, *Journal of Analytical Methods in Chemistry* 2020 (December): e8867653. ISSN: 2090-8865, accessed March 23, 2021. https://doi.org/10.1155/2020/8867653. https://www.hindawi.com/journals/jamc/2020/8867653/.

# 7 Appendices

## A Supplementary pictures from Chapter 3 Method

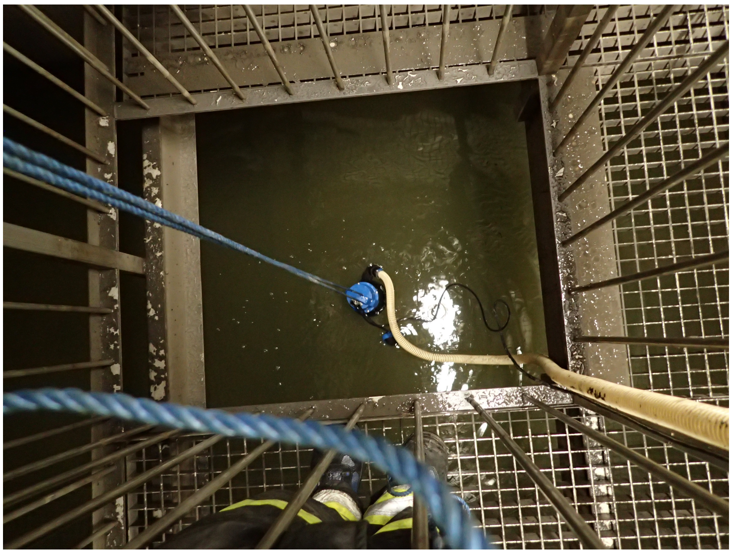


Figure 20: Submersible pump used to pump up tunnel wash water samples for this thesis.

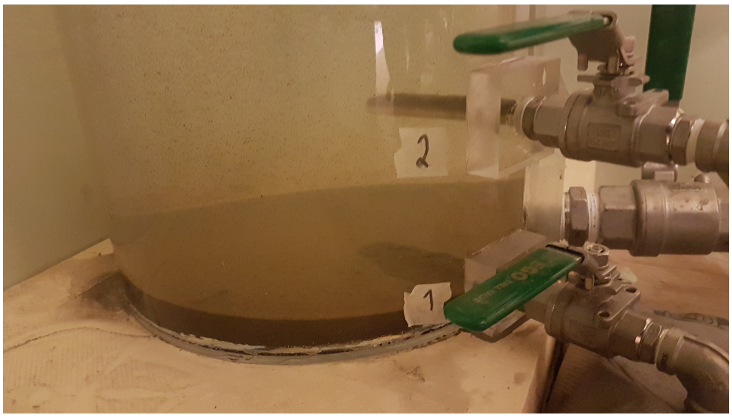


Figure 21: Sediment layer in the bottom of sedimentation column after 30 days of sedimentation, interfering with the lowermost extraction tap, marked "1" in the photo.

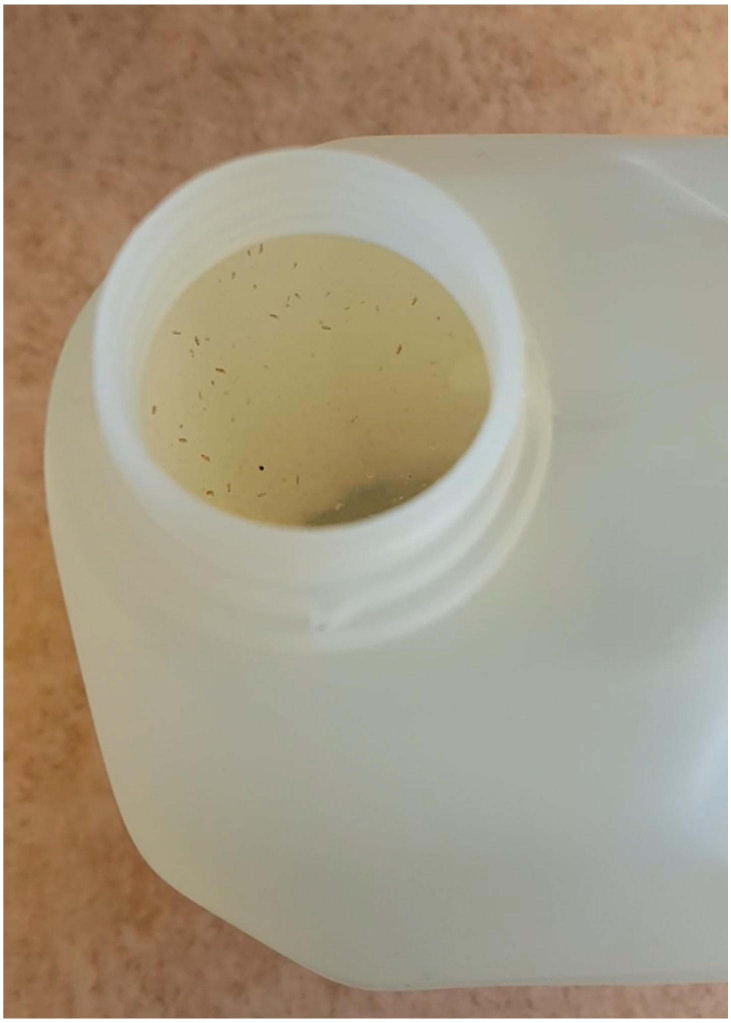


Figure 22: Living aquatic organisms sampled from the sedimentation basin of the Tåsen tunnel's eastbound tunnel tube.

# B Particle size distributions: optical and laser detection

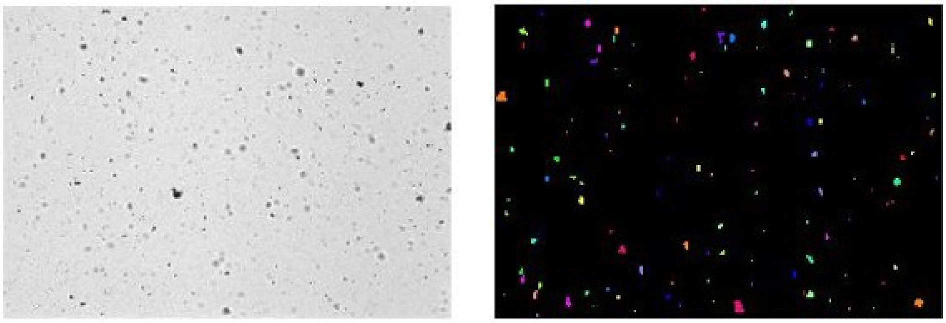


Figure 23: Picture to the left taken of untreated tunnel wash water from Smestadtunnelen (Oslo, Norway) flowing through the glass flow cell of PN3000 XPT Micro Particle Detector (Postnova). 100 such pictures were taken while continuously pumping sample through the cell. Picture to the right shows the encoded picture of detected particles. In the encoded picture, 139 particles were detected, with mean Waddel disk diameter of 8 $\mu$m and standard deviation of 4 $\mu$m.

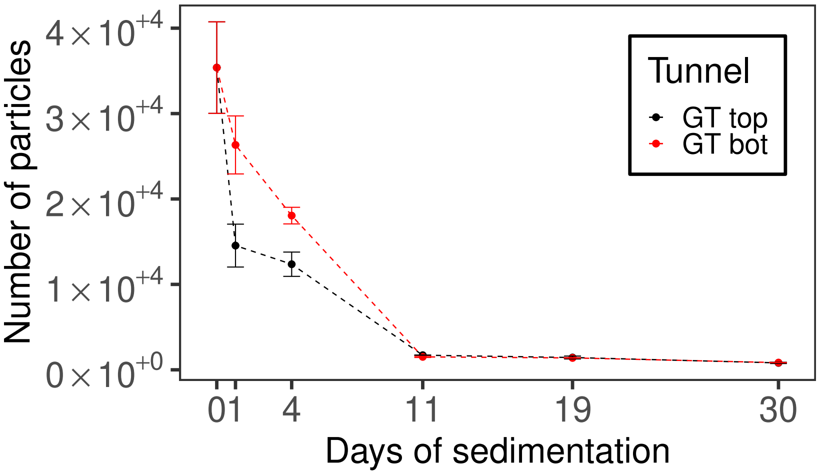
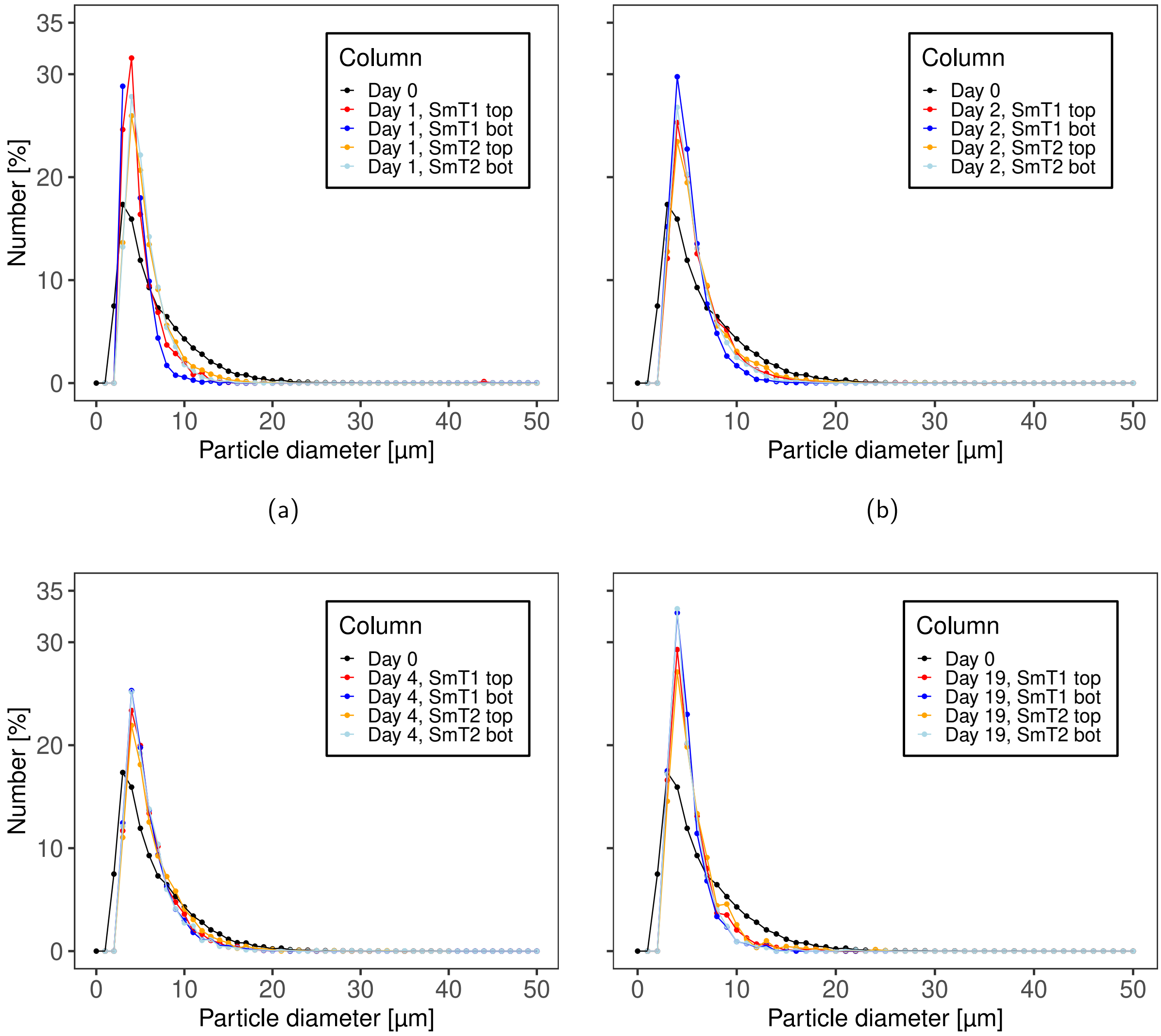


Figure 24: Mean, relative number of particles with standard deviations, a result of optical particle analyses. Mean values and standard deviations found based on measurements of two parallels. GT top, GT bot: top and bottom of two columns with sample from Grillstadtunnelen. The plot shows more rapid decrease in particle number in the top compared to the bottom until four days of sedimentation, before becoming uniform from day 11 and on-wards.



(a) (b)

(c) (d)

Figure 25: Similarities in number weighted optical particle size distribution (PSD) of tunnel wash water sampled from the Smestadtunnelen. The four sub figures illustrate the similarities between PSD in the two parallel columns SmT1 (Smestadtunnelen 1) and SmT2 (Smestadtunnelen 2). (a) One day of sedimentation (b) Two days of sedimentation (c) Four days of sedimentation (d) Nineteen days of sedimentation. Therefore, the mean PSDs ere given in the thesis.

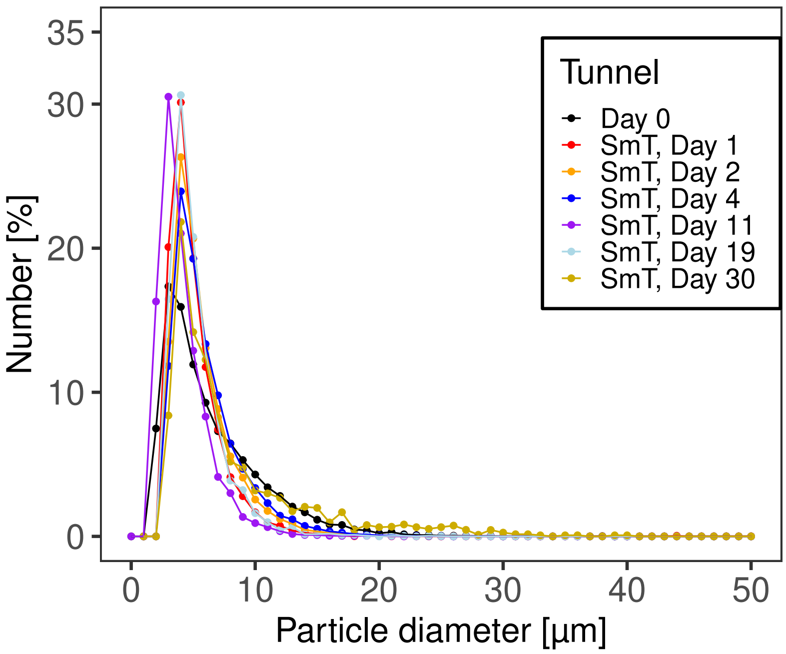


Figure 26: Smestadtunnelen (SmT): Optical particle size distribution during 30 days of sedimentation, illustrating the irregular, fluctuating tail of particles larger than 10 $\mu$m. Simultaneously, the peak of smaller particles also varies.

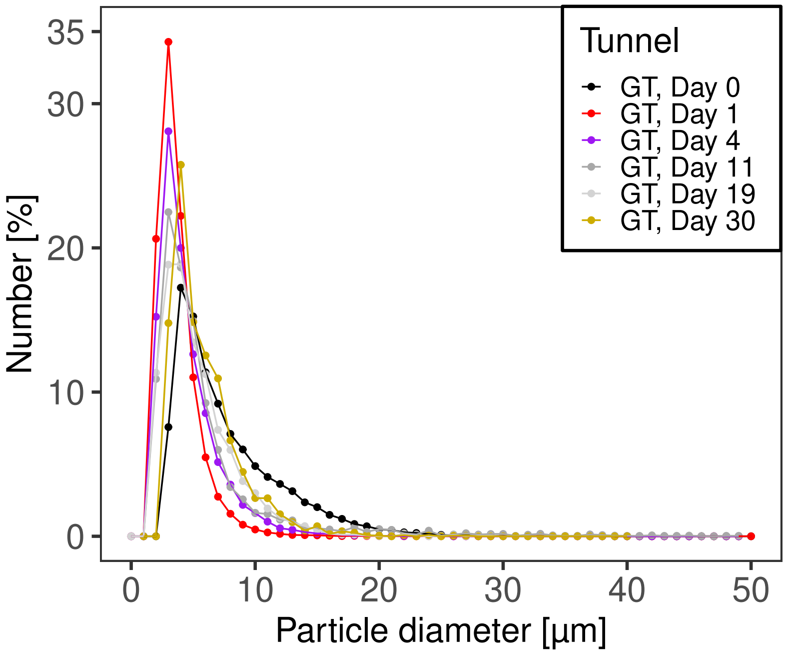


Figure 27: Grillstadtunnelen (GT): Optical particle size distribution during 30 days of sedimentation, illustrating the irregular, fluctuating tail of particles larger than 10 $\mu$m. Simultaneously, the peak of smaller particles also varies.

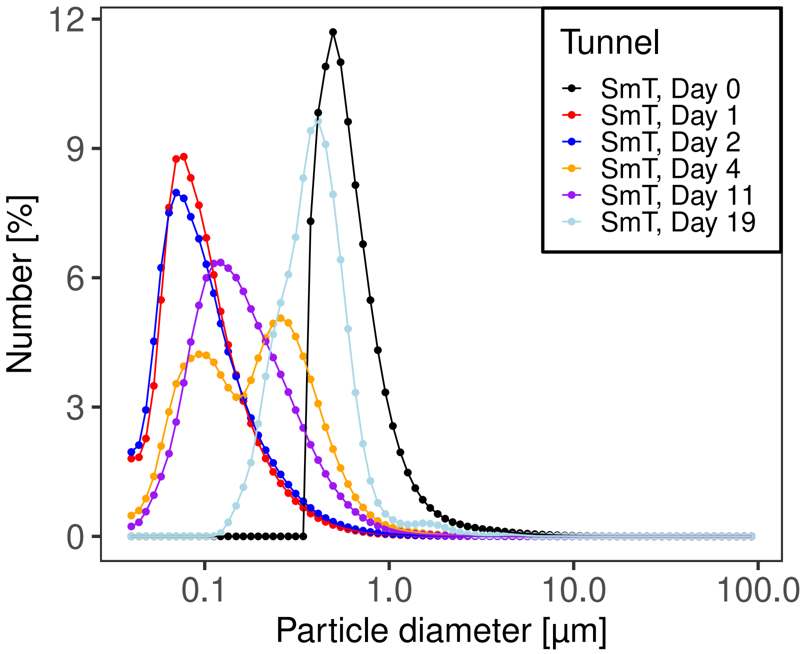


Figure 28: Smestadtunnelen (SmT): particle size distribution by laser diffraction for all measured days, illustrating how the irregular, fluctuating number-weighted peaks smaller than 0.1 $\mu$m dominate in numbers.

# C Safety data sheet for Purify Infravask detergent

The following 10 pages is the attachment of the Safety data sheet of *Purify Infravask*, which is used to wash Tåsentunnelen and Smestadtunnelen in Oslo, Norway (2021).

# SIKKERHETSDATABLAD
# PG-IV1-X1-1000

Sikkerhetsdatabladet er i samsvar med Kommisjonsforordning (EU) 2015/830 av 28 mai 2015 om endring av europaparlaments- og rådsforordning (EF) nr. 1907/2006 om registrering, vurdering, godkjenning og begrensning av kjemikalier (REACH)

## AVSNITT 1: IDENTIFIKASJON AV STOFFET / STOFFBLANDINGEN OG AV SELSKAPET / FORETAKET

| | |
|---|---|
| Utgitt dato | 19.05.2017 |

### 1.1. Produktidentifikator

| | |
|---|---|
| Kjemikaliets navn | PG-IV1-X1-1000 |
| Synonymer | Infravask |

### 1.2. Identifiserte relevante bruksområder for stoffet eller stoffblandingen og bruk som det advares mot

| | |
|---|---|
| Produktgruppe | Rengjøringsmiddel. |
| Kjemikaliets bruksområde | Til vask av tunnel, bro, skilt og infrastruktur. |

### 1.3. Opplysninger om leverandøren av sikkerhetsdatabladet

**Produsent**

| | |
|---|---|
| Firmanavn | PURIFY AS |
| Postadresse | Kanalsletta 9 |
| Postnr. | 4065 |
| Poststed | Stavanger |
| Land | Norge |
| Telefon | 97194997 |
| E-post | claudio@cobnorge.com |
| Kontaktperson | Claudio Michel |

### 1.4. Nødtelefonnummer

| | |
|---|---|
| Nødtelefon | Telefon: 22 59 13 00<br>Beskrivelse: Giftinformasjonen |

## AVSNITT 2: FAREIDENTIFIKASJON

### 2.1. Klassifisering av stoffet eller stoffblandingen

| | |
|---|---|
| Klassifisering i henhold til CLP (EC) No 1272/2008 [CLP / GHS] | Eye Dam. 1; H318 |
| Stoffets/blandingens farlige egenskaper | Gir alvorlig øyeskade. |

### 2.2. Merkingselementer

## Farepiktogrammer (CLP)

| | |
|---|---|
| Sammensetning på merkeetiketten | Benzensulfonsyre, alkylderiv., natriumsalter < 5 %, Natriumlauryletersulfat < 3 % |
| Varselord | Fare |
| Faresetninger | H318 Gir alvorlig øyeskade. |
| Sikkerhetssetninger | P101 Dersom det er nødvendig med legehjelp, ha produktets beholder eller etikett for hånden.<br>P102 Oppbevares utilgjengelig for barn.<br>P280 Benytt vernebriller/ansiktsskjerm.<br>P305+P351+P338 VED KONTAKT MED ØYNE: Skyll forsiktig med vann i flere minutter. Fjern eventuelle kontaktlinser dersom dette enkelt lar seg gjøre. Fortsett skyllingen.<br>P310 Kontakt umiddelbart et GIFTINFORMASJONSSENTER eller lege. |
| Supplerende faresetninger på etikett | Innhold i henhold til bestemmelser om vaskemidler:<br>5-15% anioniske overflateaktive stoffer. |

## 2.3. Andre farer

| | |
|---|---|
| PBT / vPvB | Blandingen oppfyller ikke gjeldende kriterier for PBT (Persistente, Bioakkumulerbare og Toksiske) eller vPvB (veldig Persistent og veldig Bioakkumulerende). |

# AVSNITT 3: SAMMENSETNING/OPPLYSNINGER OM BESTANDDELER

## 3.2. Stoffblandinger

| Komponentnavn | Identifikasjon | Klassifisering | Innhold |
|---|---|---|---|
| Benzensulfonsyre, alkylderiv., natriumsalter | CAS-nr.: 68411-30-3<br>EC-nr.: 270-115-0 | Skin Irrit. 2; H315<br>Eye Dam. 1; H318<br>Acute tox. 4; H302<br>Aquatic Chronic 3; H412 | < 5 % |
| Natriumlauryletersulfat | CAS-nr.: 9004-82-4 | Skin Irrit. 2; H315<br>Eye Dam. 1; H318<br>Aquatic Chronic 3; H412 | < 3 % |
| Innhold i henhold til bestemmelser om vaskemidler: | | | |
| Anioniske overflateaktive stoffer | | | 5 – 15 % |
| Komponentkommentarer | Se avsnitt 16 for forklaring av faresetninger (H). | | |

# AVSNITT 4: FØRSTEHJELPSTILTAK

## 4.1. Beskrivelse av førstehjelpstiltak

| | |
|---|---|
| Generelt | Nødtelefon: se avsnitt 1.4.<br>Ved bevisstløshet eller alvorlige tilfeller, ring 113. |
| Innånding | Frisk luft, ro og varme. Kontakt lege hvis ikke alt ubehag gir seg. |
| Hudkontakt | Fjern tilsølt tøy. Skyll straks med mye vann. Kontakt lege hvis ikke alt ubehag gir seg. |
| Øyekontakt | Skyll straks med store mengder vann (temperert 20-30°C) i minst 30 min. Fjern evt. kontaktlinser. Hold øyelokket åpent. Transport til lege. Fortsett skyllingen under transporten. |
| Svelging | Skyll munn med vann. Drikk et par glass vann eller melk. Fremkall ikke brekning. Kontakt lege hvis ikke alt ubehag gir seg. |

## 4.2. De viktigste symptomene og virkningene, både akutte og forsinkede

| | |
|---|---|
| Akutte symptomer og virkninger | Øyekontakt: Kjemikaliet virker etsende på øynene og kan forårsake varig skade. Symptomer som sterk svie, rennende øyne, rødhet og tåkesyn vil kunne oppstå. I alvorlige tilfeller er det fare for synsskade/blindhet.<br>Svelging: Svelging av kjemikaliet kan forårsake ubehag. |

## 4.3. Angivelse av om umiddelbar legehjelp og spesialbehandling er nødvendig

| | |
|---|---|
| Annen informasjon | Symptomatisk behandling. Ingen spesifikk informasjon fra produsent. |

# AVSNITT 5: BRANNSLOKKINGSTILTAK

## 5.1. Slokkingsmidler

| | |
|---|---|
| Egnede slokkingsmidler | Velges i forhold til omgivende brann. |
| Uegnede slokkingsmidler | Bruk ikke samlet vannstråle. |

## 5.2. Særlige farer knyttet til stoffet eller stoffblandingen

| | |
|---|---|
| Brann- og eksplosjonsfarer | Kjemikaliet er ikke brennbart. |
| Farlige forbrenningsprodukter | Kan inkludere, men er ikke begrenset til: Karbondioksid (CO2). Karbonmonoksid (CO). Svovelholdige gasser (SOx). Uspesifiserte organiske forbindelser. |

## 5.3. Råd til brannmannskaper

| | |
|---|---|
| Personlig verneutstyr | Bruk trykkluftmaske når kjemikaliet er involvert i brann. Ved rømning brukes godkjent rømningsmaske. Se forøvrig avsnitt 8. |
| Annen informasjon | Beholdere i nærheten av brann flyttes straks eller kjøles med vann. |

# AVSNITT 6: TILTAK VED UTILSIKTET UTSLIPP

## 6.1. Personlige forsiktighetsregler, personlig verneutstyr og nødrutiner

| | |
|---|---|
| Sikkerhetstiltak for å beskytte personell | Benytt personlig verneutstyr som angitt i avsnitt 8.<br>Sørg for tilstrekkelig ventilasjon.<br>Unngå kontakt med øynene.<br>Ved søl: Vær oppmerksom på glatte gulv og overflater. |

## 6.2. Forsiktighetsregler med hensyn til miljø

| | |
|---|---|
| Sikkerhetstiltak for å beskytte ytre miljø | Forhindre utslipp til kloakk, vassdrag eller grunn. |

## 6.3. Metoder og materialer for oppsamling og rensing

| | |
|---|---|
| Opprydding | Absorber i vermikulitt, tørr sand eller jord og fyll i beholdere. Spill samles opp i egnede beholdere og leveres til destruksjon som avfall iht. avsnitt 13. |

## 6.4. Henvisning til andre avsnitt

| | |
|---|---|
| Andre anvisninger | Se også avsnitt 8 og 13. |

# AVSNITT 7: HÅNDTERING OG LAGRING

## 7.1. Forsiktighetsregler for sikker håndtering

| | |
|---|---|
| Håndtering | Bruk angitt verneutstyr, se avsnitt 8.<br>Sørg for tilstrekkelig ventilasjon. |

| | |
|---|---|
| | Unngå kontakt med øynene.<br>Spill gjør gulv og arbeidsredskaper glatte og sleipe. |

## Beskyttelsestiltak

| | |
|---|---|
| Råd om generell yrkeshygiene | Det må ikke spises, drikkes eller røykes under arbeidet. Vask hendene etter hvert skift og før spising, røyking eller bruk av toalett. Vask tilsølte klær før de brukes. |

## 7.2. Vilkår for sikker lagring, herunder eventuelle uforenligheter

| | |
|---|---|
| Oppbevaring | Oppbevares i tett lukket beholder på et tørt sted. |

## Betingelser for sikker oppbevaring

| | |
|---|---|
| Råd angående samlagring | Oppbevares adskilt fra næringsmidler. |
| Lagringstemperatur | Verdi: < 40 °C |

## 7.3. Særlig(e) sluttanvendelse(r)

| | |
|---|---|
| Spesielle bruksområder | Se avsnitt 1.2. |

# AVSNITT 8: EKSPONERINGSKONTROLL/PERSONBESKYTTELSE

## 8.1. Kontrollparametere

| | |
|---|---|
| Annen informasjon om grenseverdier | Inneholder ingen stoffer med grenseverdi for forurensninger i arbeidsatmosfæren.<br>Referanser (lover/forskrifter): FOR 2011-12-06 nr 1358 Forskrift om tiltaks– og grenseverdier (sist endret gjennom FOR-2016-12-22-1860). |

## 8.2. Eksponeringskontroll

## Forholdsregler for å hindre eksponering

| | |
|---|---|
| Tekniske tiltak for å hindre eksponering | Sørg for tilstrekkelig ventilasjon. Personlig verneutstyr skal være CE-merket og bør velges i samarbeid med leverandøren av slikt utstyr. Det anbefalte verneutstyret og de angitte standardene er veiledende. Standarder skal være av nyeste versjon.<br>Risikovurdering av den aktuelle arbeidsplassen/-operasjonen (faktisk risiko) kan medføre andre vernetiltak. Verneutstyrets egnethet og slitestyrke vil avhenge av bruksområde. |

## Øye- / ansiktsvern

| | |
|---|---|
| Øyevernutstyr | Beskrivelse: Bruk tettsittende vernebriller eller ansiktsskjerm.<br>Referanser til relevante standarder: NS-EN 166 (Øyevern – Spesifikasjoner). |
| Ytterligere øyeverntiltak | Øyedusj skal være på arbeidsplassen. Enten en fast øyedusjenhet koblet til drikkevann (temperert vann ønskelig) eller en bærbar disponibel enhet (øyespyleflaske). |

## Håndvern

| | |
|---|---|
| Egnede materialer | Neopren, nitril, polyetylen eller PVC. |
| Gjennomtrengningstid | Kommentarer: Ingen spesifikk informasjon fra produsent. |
| Tykkelsen av hanskemateriale | Kommentarer: Ingen spesifikk informasjon fra produsent. |
| Håndvernsutstyr | Beskrivelse: Benytt hansker av motstandsdyktig materiale. Det angitte hanskematerialet er foreslått etter en gjennomgang av enkeltstoffene i kjemikaliet og kjente hanskeguider. Hansketykkelse må velges i samarbeid med hanskeleverandøren, som kan opplyse om hanskematerialets gjennomtrengningstid. Hanskenes egenskaper kan variere hos de ulike hanskeprodusentene.<br>Referanser til relevante standarder: NS-EN 374 (Vernehansker mot kjemikalier og mikroorganismer).<br>NS-EN 420 (Vernehansker – Generelle krav og prøvingsmetoder). |

| | |
|---|---|
| Ytterligere håndbeskyttelsestiltak | Skift hansker ved tegn på slitasje. |

## Hudvern

| | |
|---|---|
| Anbefalte verneklær | Beskrivelse: Benytt hensiktsmessige verneklær for beskyttelse mot hudkontakt. |
| Ytterligere hudbeskyttelsestiltak | Nøddusj skal være tilgjengelig på arbeidsplassen. |

## Åndedrettsvern

| | |
|---|---|
| Anbefalt åndedrettsvern | Beskrivelse: Normalt ikke nødvendig. Ved utilstrekkelig ventilasjon brukes maske med filter A mot løsemiddeldamper.<br>Referanser til relevante standarder: NS-EN 14387 (Åndedrettsvern – Gassfiltre og kombinerte filtre – Krav, prøving, merking). |

## Passende miljømessig eksponeringskontroll

| | |
|---|---|
| Begrensning av miljøeksponering | Forhindre utslipp til kloakk, vassdrag eller grunn. |

# AVSNITT 9: FYSISKE OG KJEMISKE EGENSKAPER

## 9.1. Opplysninger om grunnleggende fysiske og kjemiske egenskaper

| | |
|---|---|
| Tilstandsform | Væske |
| Farge | Ikke angitt av produsenten. |
| Lukt | Ikke angitt av produsenten. |
| Luktgrense | Kommentarer: Ikke angitt av produsenten. |
| pH | Status: I handelsvare<br>Kommentarer: Ikke angitt av produsenten. |
| Smeltepunkt / smeltepunktintervall | Kommentarer: Ikke angitt av produsenten. |
| Kokepunkt / kokepunktintervall | Kommentarer: Ikke angitt av produsenten. |
| Flammepunkt | Kommentarer: Ikke angitt av produsenten. |
| Fordampningshastighet | Kommentarer: Ikke angitt av produsenten. |
| Antennelighet (fast stoff, gass) | Ikke angitt av produsenten. |
| Eksplosjonsgrense | Kommentarer: Ikke angitt av produsenten. |
| Damptrykk | Kommentarer: Ikke angitt av produsenten. |
| Damptetthet | Kommentarer: Ikke angitt av produsenten. |
| Relativ tetthet | Kommentarer: Ikke angitt av produsenten. |
| Løslighet | Medium: Vann<br>Kommentarer: Løselig. |
| Fordelingskoeffisient: n-oktanol/vann | Kommentarer: Ikke angitt av produsenten. |
| Selvantennelighet | Kommentarer: Ikke angitt av produsenten. |
| Dekomponeringstemperatur | Kommentarer: Ikke angitt av produsenten. |
| Viskositet | Kommentarer: Ikke angitt av produsenten. |
| Eksplosive egenskaper | Ikke angitt av produsenten. |
| Oksiderende egenskaper | Ikke angitt av produsenten. |

## 9.2. Andre opplysninger

## Andre fysiske og kjemiske egenskaper

| | |
|---|---|
| Kommentarer | Ingen ytterligere informasjon er tilgjengelig. |

# AVSNITT 10: STABILITET OG REAKTIVITET

## 10.1. Reaktivitet

| | |
|---|---|
| Reaktivitet | Ikke angitt av produsenten. |

## 10.2. Kjemisk stabilitet

| | |
|---|---|
| Stabilitet | Ikke angitt av produsenten. |

## 10.3. Mulighet for farlige reaksjoner

| | |
|---|---|
| Risiko for farlige reaksjoner | Ikke angitt av produsenten. |

## 10.4. Forhold som skal unngås

| | |
|---|---|
| Forhold som skal unngås | Ikke angitt av produsenten. |

## 10.5. Uforenlige materialer

| | |
|---|---|
| Materialer som skal unngås | Ikke angitt av produsenten. |

## 10.6. Farlige nedbrytningsprodukter

| | |
|---|---|
| Farlige spaltningsprodukter | Ingen under normale forhold. Se også avsnitt 5.2. |

## Annen informasjon

# AVSNITT 11: TOKSIKOLOGISKE OPPLYSNINGER

## 11.1. Opplysninger om toksikologiske virkninger

## Øvrige helsefareopplysninger

| | |
|---|---|
| Vurdering av akutt toksisitet klassifisering | Kriteriene for klassifisering kan på grunnlag av de foreliggende data ikke anses for å være oppfylt. |
| Vurdering hudetsende / hudirriterende, klassifisering | Kriteriene for klassifisering kan på grunnlag av de foreliggende data ikke anses for å være oppfylt. |
| Vurdering øyeskade / øyeirritasjon, klassifisering | Gir alvorlig øyeskade. |
| Vurdering av luftveissensibilisering, klassifisering | Kriteriene for klassifisering er på grunnlag av de tilgjengelige data ikke ansett å være oppfylt. |
| Vurdering av hudsensibilisering, klassifisering | Kriteriene for klassifisering er på grunnlag av de tilgjengelige data ikke ansett å være oppfylt. |
| Innånding | Ingen spesifikk informasjon fra produsent. |
| Hudkontakt | Ingen hudirritasjon forventes. |
| Øyekontakt | Kjemikaliet er sterkt etsende på øynene og kan forårsake varig skade. Symptomer som sterk svie, rennende øyne, rødhet og tåkesyn vil kunne oppstå. I alvorlige tilfeller er det fare for synsskade/blindhet. |
| Svelging | Kan forårsake ubehag ved svelging. |

| | |
|---|---|
| Vurdering av arvestoffskadelig virkning på kjønnsceller, klassifisering | Kriteriene for klassifisering er på grunnlag av de tilgjengelige data ikke ansett å være oppfylt. |
| Vurdering kreftfremkallende egenskaper, klassifisering | Kriteriene for klassifisering er på grunnlag av de tilgjengelige data ikke ansett å være oppfylt. |
| Vurdering av reproduksjonstoksisitet, Klassifisering | Kriteriene for klassifisering er på grunnlag av de tilgjengelige data ikke ansett å være oppfylt. |
| Vurdering av bestemt målorgan SE, klassifisering | Kriteriene for klassifisering er på grunnlag av de tilgjengelige data ikke ansett å være oppfylt. |
| Vurdering av bestemt målorgan RE, klassifisering | Kriteriene for klassifisering er på grunnlag av de tilgjengelige data ikke ansett å være oppfylt. |
| Vurdering aspirasjonsfare klassifisering | Kriteriene for klassifisering kan på grunnlag av de foreliggende data ikke anses for å være oppfylt. |

# AVSNITT 12: ØKOLOGISKE OPPLYSNINGER

## 12.1. Giftighet

| | |
|---|---|
| Økotoksisitet | Kjemikaliet er ikke klassifisert som miljøskadelig. |

## 12.2. Persistens og nedbrytbarhet

| | |
|---|---|
| Persistens og nedbrytbarhet | Tensidet(ene) som inngår i denne blandingen oppfyller kriteriene for biologisk nedbrytning i EU regulativ nr. 648/2004 som omhandler vaske– og rengjøringsmidler. |

## 12.3. Bioakkumuleringsevne

| | |
|---|---|
| Kommentarer, Bioakkumulering | Ingen spesifikk informasjon fra produsent. |

## 12.4. Mobilitet i jord

| | |
|---|---|
| Mobilitet | Løselig i vann. Kan spres i jord og grunnvann. |

## 12.5. Resultater av PBT og vPvB vurdering

| | |
|---|---|
| PBT vurderingsresultat | Blandingen oppfyller ikke gjeldende kriterier for PBT (Persistente, Bioakkumulerbare og Toksiske). |
| vPvB vurderingsresultat | Blandingen oppfyller ikke gjeldende kriterier for vPvB (veldig Persistent og veldig Bioakkumulerende). |

## 12.6. Andre skadevirkninger

| | |
|---|---|
| Andre skadevirkninger / annen informasjon | Forhindre utslipp til kloakk, vassdrag eller grunn. |

# AVSNITT 13: DISPONERING

## 13.1. Avfallsbehandlingsmetoder

| | |
|---|---|
| Egnede metoder til fjerning av kjemikaliet | Leveres som farlig avfall til godkjent behandler eller innsamler. Koden for farlig avfall (EAL-kode) er veiledende. Bruker må selv angi riktig EAL-kode hvis bruksområdet avviker. |
| Avfallskode EAL | Avfallskode EAL: 07 06 99 avfall som ikke er spesifisert andre steder<br>Klassifisert som farlig avfall: Nei |
| Annen informasjon | Må ikke helles i avløp. |

# AVSNITT 14: TRANSPORTOPPLYSNINGER

## 14.1. UN-nummer

| | |
|---|---|
| Kommentar | Ikke farlig i forbindelse med transport under UN, IMO, ADR/RID og IATA/ICAO regler. |

## 14.2. FN-forsendelsesnavn

| | |
|---|---|
| Kommentar | Ikke relevant. |

## 14.3. Transportfareklasse(r)

| | |
|---|---|
| Kommentar | Ikke relevant. |

## 14.4. Emballasjegruppe

| | |
|---|---|
| Kommentar | Ikke relevant. |

## 14.5. Miljøfarer

| | |
|---|---|
| Kommentar | Ikke relevant. |

## 14.6. Særlige forsiktighetsregler ved bruk

| | |
|---|---|
| Spesielle forholdsregler | Ikke relevant. |

## 14.7. Bulktransport i henhold til vedlegg II i MARPOL 73/78 og IBC-regelverket

| | |
|---|---|
| Forurensningskategori | Ikke relevant. |

## Andre relevante opplysninger

| | |
|---|---|
| Andre relevante opplysninger | Ikke relevant. |

# AVSNITT 15: OPPLYSNINGER OM BESTEMMELSER

## 15.1. Særlige bestemmelser/særskilt lovgivning om sikkerhet, helse og miljø for stoffet eller stoffblandingen

| | |
|---|---|
| Referanser (Lover/Forskrifter) | Forskrift om klassifisering, merking og emballering av stoffer og stoffblandinger (CLP) av 16.06.2012 med senere endringer.<br>Forskrift om registrering, vurdering, godkjenning og begrensning av kjemikalier (REACH-forskriften) av 30. mai 2008 med senere endringer.<br>FOR 2009-04-01 nr 384: Forskrift om landtransport av farlig gods med senere endringer, Direktoratet for samfunnssikkerhet og beredskap.<br>FOR 2004-06-01 nr. 922: Forskrift om begrensning i bruk av helse– og miljøfarlige kjemikalier og andre produkter (produktforskriften), med senere endringer; §§2-12, 2-14, Vaskemidler.<br>Avfallsforskriften, FOR 2004-06-01 nr 930, fra Miljøverndepartementet. |

## 15.2. Vurdering av kjemikaliesikkerhet

| | |
|---|---|
| Vurdering av kjemikaliesikkerhet er gjennomført | Nei |

# AVSNITT 16: ANDRE OPPLYSNINGER

| | |
|---|---|
| Leverandørens anmerkninger | Informasjonen i dette dokument skal gjøres tilgjengelig for alle som håndterer kjemikaliet. |
| Liste over relevante H-setninger (i avsnitt 2 og 3). | H302 Farlig ved svelging.<br>H315 Irriterer huden. |

| | |
|---|---|
| | H318 Gir alvorlig øyeskade.<br>H412 Skadelig, med langtidsvirkning, for liv i vann. |
| Klassifisering i henhold til CLP (EC) No 1272/2008 [CLP / GHS] | Eye Dam. 1; H318 |
| Viktige litteraturreferanser og datakilder | Sikkerhetsdatabladet er utarbeidet med basis i opplysninger gitt av produsenten. |
| Brukte forkortelser og akronymer | PBT: Persistent, Bioakkumulerende og Toksisk (giftig)<br>vPvB: veldig Persistent og veldig Bioakkumulerende<br>ADR: The European Agreement concerning the International Carriage of Dangerous Goods by Road<br>RID: The Regulations concerning the International Carriage of Dangerous Goods by Rail<br>ICAO: The International Civil Aviation Organisation<br>IATA: The International Air Transport Association<br>IMDG: The International Maritime Dangerous Goods Code<br>EAL-kode: kode fra EUs felles klassifiseringssystem for avfall (EWC = European Waste Code)<br>OECD: Organisation for Economic Cooperation and Development. |
| Opplysninger som er nye, slettet eller revidert | Nytt sikkerhetsdatablad |
| Kvalitetssikring av informasjonen | Dette sikkerhetsdatablad er kvalitetskontrollert av Kiwa Teknologisk Institutt as, som er sertifisert iht. ISO 9001:2008. |
| Utarbeidet av | Kiwa Teknologisk Institutt as v/ Hanna M. Storrvik |

# D Safety data sheet for Sanego Solvynol Truck detergent

The following 7 pages is the attachment of the safety data sheet of *Sanego Solvynol Truck*, which is used to wash Grillstadtunnelen and Strindheimtunnelen in Trondheim, Norway (2021).

# SIKKERHETSDATABLAD

I samsvar med 453/2010 og 1272/2008

(Alle henvisninger til EUs regelverk og direktiver er forkortet til bare nummerbetegnelse)

Utgitt 2016-03-10

## AVSNITT 1: IDENTIFIKASJON AV STOFFET/STOFFBLANDINGEN OG AV SELSKAPET/FORETAKET

### 1.1. Produktidentifikator

| | |
|---|---|
| **Handelsnavn** | **SOLVYNOL TRUCK** |

### 1.2. Identifiserte relevante bruksområder for stoffet eller stoffblandingen og bruk som det advares mot

| | |
|---|---|
| **Identifiserte bruksområder** | Rengjøringsmiddel |

### 1.3. Opplysninger om leverandøren av sikkerhetsdatabladet

| | |
|---|---|
| **Firma** | SANEGO AB<br>Vallevägen 26<br>SE-37300 JÄMJÖ<br>Sverige |
| **Telefon** | +46(0)455-566 40 |
| **E-post** | info@sanego.se |
| **Nettområde** | www.sanego.se |

Offisiell norsk oversettelse av forordning 453/2010 foreligger ikke til dags dato. Ifølge melding fra Klima- og forurensningsdirektoratets REACH-hjelp benyttes overskrifter og underoverskrifter fra svensk utgave (markert med *). Endringer gjøres ved første oppdatering etter at offisiell norsk oversettelse foreligger.

### 1.4. Nødtelefonnummer

I nødsfall, kontakt giftinformasjon: 22 59 13 00; Medisinsk nødhjelp: tel 113; brann: 110; Politi: 112

Ikke akutt giftinformasjon: https://helsenorge.no/Giftinformasjon

## AVSNITT 2: FAREIDENTIFIKASJON

### 2.1. Klassifisering av stoffet eller stoffblandingen

**Klassifisering i henhold til 1272/2008**

Etsende (kategori 1B)

**Klassifisering i henhold til 1999/45/EG**

irriterende; Xi; R38 R41.

### 2.2. Merkingselementer

**Etikettinformasjon i henhold til 1272/2008**

Farepiktogrammer

| | |
|---|---|
| Signalord | Fare |
| Faresetninger | |
| H314 | Gir alvorlige etseskader på hud og øyne |
| Sikkerhetssetninger | |
| P280 | Benytt vernehansker og vernebriller |
| P301+330+331 | Benytt vernehandskerr og skyddsklaeder og vernebriller |
| P303+P361+P353 | VED HUDKONTAKT (eller håret): Tilsølte klær må fjernes straks. Skyll/dusj huden med vann |
| P305+P351+P338 | VED KONTAKT MED ØYNENE: Skyll forsiktig med vann i flere minutter. Fjern eventuelle kontaktlinser dersom dette enkelt lar seg gjøre. Fortsett skyllingen |
| P310 | Kontakt umiddelbart et GIFTINFORMASJONSSENTER |

### Etikettinformasjon i henhold til 1999/45/EG

Se Avsnitt 16.

## 2.3 Andre farer

Ikke aktuelt.

# AVSNITT 3: SAMMENSETNING/OPPLYSNINGER OM BESTANDDELER

Dette produktet består av en homogen vannløsning.

## 3.2. Stoffblandinger

Merk at tabellen viser kjente farer for ingrediensene i ren form. Farene reduseres eller elimineres når de blandes eller spes ut, se Avsnitt 16d.

| Bestanddeler | Klassifisering | Konsentrasjon |
|---|---|---|
| **FETTALKOHOLETOKSYLAT** | | |
| CAS-Nr 160875-66-1 | Acute Tox 4*oral*, Eye Dam 1; H302, H318 | 1-5% |
| EG-nummer 605-233-7 | Xn; R22 R41 | |
| **FATTY ALCOHOL ALKOXYLATED** | | |
| CAS-Nr 166736-08-9 | Acute Tox 4*oral*, Eye Dam 1; H302, H318 | 1-5% |
| EG-nummer 605-450-7 | Xn; R22 R41 | |
| | ; - | |
| **NATRIUMHYDROKSID** | | |
| CAS-Nr 1310-73-2 | Skin Corr 1A; H314 | 1-5% |
| EG-nummer 215-185-5 | C; R35 | |
| Indeksnummer 011-002-00-6 | | |
| **KOKOSAMIDOPROPYLBETAIN** | | |
| Cas nr 147170-44-3 | Eye Dam 1, H318 | 0-1 % |
| EG-nummer 931-333-8 | | |
| Reach 01-2119489410-39 | | |

Forklaringer til ingrediensene og merkingen er angitt i Avsnitt 16e. Offisielle forkortelser er skrevet med normal stil. Med kursiv stil angis spesifikasjoner og/eller kompletteringer som har blitt brukt ved beregning av blandingens klassifisering, se Avsnitt 16b.

Inneholder også komponent(er) som ikke er merkingspliktig(e).

Innhold (EF) nr. 648/2004:

< 5 % ikke-ioniske overflateaktive stoffer

< 5 % amfoterte overflateaktive stoffer.

# AVSNITT 4: FØRSTEHJELPSTILTAK

## 4.1. Beskrivelse av førstehjelpstiltak

### Generelt

Kontakt umiddelbart et GIFTINFORMASJONSSENTER eller lege.

### Ved innånding

La den skadede hvile på varm plass med frisk luft. Kontakt lege.

### Ved øyekontakt

Om mulig, ta øyeblikkelig ut eventuelle kontaktlinser.

Skyl øjet i flere minutter med tempereret vand. Kontakt lege.

### Ved hudkontakt

Vask med rikelig med vann og kontakt lege.

Ta av forurensede klær.

### Ved svelging

Skyll først munnen nøye med mye vann men SVELG IKKE; Drikk minst en halv liter vann og kontakt lege. IKKE brekninger.

## 4.2. De viktigste symptomene og virkningene, både akutte og forsinkede

Etseskader kan oppstå.

## 4.3. Angivelse av om umiddelbar legehjelp og spesialbehandling er nødvendig

Symptomatisk behandling.

# AVSNITT 5: BRANNSLOKKINGSTILTAK

### 5.1. Slokkingsmidler

Slokkes med middel avsett for omgivende brann.

### 5.2. Særlige farer knyttet til stoffet eller stoffblandingen

Ved brann kan etsende gasser spres.

Merk at slukkevannet kan være etsende.

Ikke brennbart.

### 5.3. Råd til brannmannskaper

Ved brann, bruk dress, som beskytter mot etsende stoffer.

Ved brann, bruk uavhengig pusteapparat.

# AVSNITT 6: TILTAK VED UTILSIKTET UTSLIPP

### 6.1. Personlige forsiktighetsregler, personlig verneutstyr og nødrutiner

Inhaler ikke dampen og unnvik kontakt med hud, øyne og klær.

Bruk beskyttelseshansker som beskytter mot etsende stoffer.

Beskytt ansikt og øyne med visir eller vernebriller.

Merk at det er fare for å gli dersom produktet lekker/søles.

### 6.2. Forsiktighetsregler med hensyn til miljø

Innehold utslipp slik at den ikke kommer inn i brønner eller i jorden.

Unngå utslipp til jord, vann eller avløp.

### 6.3. Metoder og materialer for oppsamling og rensing

Suge opp væsken i inert materiale f. eks. vermikulitt, samle siden stoffet til disposisjon.

Gjør rent med vann.

Rester etter sanering er farlig avfall. Kontakt kommunens sanitære tjeneste for detaljer. Vis dette sikkerhetsdatablad.

### 6.4. Henvisning til andre avsnitt

Se avsnittene 8 og 13 for personlig verneutstyr og avfallshåndtering.

# AVSNITT 7: HÅNDTERING OG LAGRING

### 7.1. Forsiktighetsregler for sikker håndtering

Produktet ska oppbevares slik at risiko for menneskers helse eller miljøet forhindres. Unngå kontakt med mennesker og dyr og slipp ikke produktet i følsom miljø.

Hold dette produktet separert fra mat og utilgjengelig for barn og kjæledyr.

Ikke spis, drikk og røyk i rommet der dette produktet håndteres.

Arbeid slik at spill blir forhindret. Hvis spill forekommer se Avsnitt 6 i dette sikkerhetsdatablad.

### 7.2. Vilkår for sikker lagring, herunder eventuelle uforenligheter

Oppbevares tørt og kaldt (frostfritt, men ikke over 30°C).

Sørg for god ventilasjon.

Oppbevares på betryggende måte, slik at produktet ikke slipper ut i naturen.

Evakueringsplan bør være tilgjengelig, og rømningsveier må ikke blokkeres.

Nøddusj og øyneskylling skal finnes på arbeidsplassen.

Oppbevares kun i originalforpakning.

### 7.3. Særlig(e) sluttanvendelse(r)

Ikke aktuelt.

# AVSNITT 8: EKSPONERINGSKONTROLL/PERSONBESKYTTELSE

### 8.1. Kontrollparametere

#### 8.1.1 Nasjonale grenseverdier, Norge

Alle ingredienser (se pkt. 3) savner hygieniske grenseverdier.

### 8.2. Eksponeringskontroll

For forebygging av yrkesrisiko må ta hensyn til helsefareerne (se punkt 2 og 10) med dette produktet i samsvar med EØS-direktiv 89/391 og 98/24 og nasjonal arbeidslovgivning.

kontaktlinser må ikke brukes vid arbeide med dette stoff.

Bruk visir eller vernebriller.

Bruk vernehansker av butylgummi, Viton eller fluorgummi, eller konsultere arbeidsmedisinsk ekspert for alternative materiale. Vis dette sikkerhetsdateblad.

Arbeid uten vernehansker bør kun skje ved håndtering av svært små mengder.

Velg mekanisk slitestyrke med hensyn til arbeidsoppgavens type i henhold til merkingen med tilsvarende piktogram med fire sifre som viser motstand mot oppskraping, kutting, opprivning og punktering, der 1 er verst og 4 eller 5 er best.

Benytt hensiktsmessige verneklær for beskyttelse ved mulig hudkontakt.

Gassmaske med filter B (grå, uorganiske gasser og damper) kan behøves.

For begrensning av miljøeksponering, se Avsnitt 12.

## AVSNITT 9: FYSISKE OG KJEMISKE EGENSKAPER

### 9.1. Opplysninger om grunnleggende fysiske og kjemiske egenskaper

| | | |
|---|---|---|
| a) | Utseende | Form: væske<br>Farge: Ingen farve |
| b) | Lukt | Ikke parfymert |
| c) | Luktterskel | Ikke aktuelt |
| d) | pH | 13-14 |
| e) | Smeltepunkt/frysepunkt | Ikke aktuelt |
| f) | Startkokepunkt og kokeområde | Ikke aktuelt |
| g) | Flammepunkt | Ikke aktuelt |
| h) | Fordampingshastighet | Ikke aktuelt |
| i) | Antennelighet (fast stoff, gass) | Ikke aktuelt |
| j) | Øvre/nedre antennelighets- eller eksplosjonsgrense | Ikke aktuelt |
| k) | Damptrykk | Ikke aktuelt |
| l) | Damptetthet | Ikke aktuelt |
| m) | Relativ tetthet | 1.08 kg/L |
| n) | Løselighet(er) | Vannløselighet Løselig |
| o) | Fordelingskoeffisient; N-oktanol/vann | Ikke aktuelt |
| p) | Selvantenningstemperatur | Ikke aktuelt |
| q) | Nedbrytingstemperatur | Ikke aktuelt |
| r) | Viskositet | Ikke aktuelt |
| s) | Eksplosjonsegenskaper | Ikke aktuelt |
| t) | Oksidasjonsegenskaper | Ikke aktuelt |

### 9.2. Andre opplysninger

Ingen informasjon tilgjengelig

## AVSNITT 10: STABILITET OG REAKTIVITET

### 10.1. Reaktivitet

Produktet inneholder ingen stoffer som kan forårsake farlige reaksjoner under normale håndterings- og bruksforhold.

### 10.2. Kjemisk stabilitet

Produktet er stabilt under normale lagrings- og bruksforhold.

### 10.3. Mulighet for farlige reaksjoner

Ingen kjente farlige reaksjoner.

### 10.4. Forhold som skal unngås

Unngå frost.

Beskyttes mot varme.

### 10.5. Uforenlige materialer

Unngå kontakt med syrer.

### 10.6. Farlige nedbrytingsprodukter

Ingen ved normale forhold.

# AVSNITT 11: TOKSIKOLOGISKE OPPLYSNINGER

### 11.1. Opplysninger om toksikologiske virkninger

**Helseskadelighet**

Produktet er ikke klassifisert som helseskadelig.

**Etsende og irriterende effekter**

Kontakt med øyne kan forårsake irreversible øyeskader.

Produktet mistenkes å kunne være etsende på hud på grunn av den høye pH-verdien.

**Relevant toksikologiske egenskaper**

**NATRIUMHYDROKSID**

LD50 kanin (Oral) 24h > 125 mg/kg oral

LD50 rotte (Oral) 24h > 500 mg/kg

# AVSNITT 12: ØKOLOGISKE OPPLYSNINGER

### 12.1. Giftighet

**NATRIUMHYDROKSID**

LC50 Stor dafnie (Daphnia magna) 48h = 30 mg/L

EC50 Stor dafnie (Daphnia magna) 48 h = 100 mg/l

LC50 Fisk 96h = 125 mg/L

LC50 solabbor (Lepomis macrochirus) 48h = 99 mg/L

IC50 Alger 72h = 10 mg/l

Ingen økologiske skader er kjent eller forventet ved normal bruk.

### 12.2. Persistens og nedbrytbarhet

Tensidene i dette produktet følger kriteriene for biologisk nedbrytbarhet i henhold til direktiv 648/2004/EF.

### 12.3. Bioakkumuleringsevne

Opplysninger om bioakkumulering mangler.

### 12.4. Mobilitet i jord

Opplysninger om bevegelighet i naturen mangler.

### 12.5. Resultater av PBT- og vPvB-vurdering

Ingen kjemikaliesikkerhetsrapport har blitt utført.

### 12.6. Andre skadevirkninger

Produktet er alkalisk og kan høyne pH-verdien lokalt ved utslipp til vann.

# AVSNITT 13: DISPONERING

### 13.1. Avfallsbehandlingsmetoder

**Avfallshåndtering for produktet**

Produktet er etsende og avfall skal, hvis den ikke behandles for å eliminere denne risikoen, anses som farlig gods.

Ta også hensyn til lokale regler for avfallshåndtering.

Forhindre utslipp av ufortynnet produkt i avløp.

Se også Avfallsforskriften (FOR-2004-06-01-930).

**Gjenvinning av produktet**

Resterende/gammelt/forurenset produkt overlates til avfallsforbrenning. Tom emballasje overlates til gjenvinning der det er praktisk mulig. Produsenten er medlem av REPA.

**Transport av avfallet**

Ikke angitt

# AVSNITT 14: TRANSPORTOPPLYSNINGER

Dette produktet forventes kun å transporteres på vei eller med tog og er derfor kun vurdert ifølge regelverkene ADR/RID. Skulle annen transportmetode bli aktuell, ta kontakt med utgiveren av dette sikkerhetsdatabladet.

### 14.1. FN-nummer

1824

### 14.2. FN-forsendelsesnavn

Natriumhydroxidlösning

### 14.3. Transportfareklasse(r)

Klass 8, Etsende

### 14.4 Emballasjegruppe

Emballasjegruppe II

### 14.5 Miljøfarer

Ikke aktuelt

### 14.6. Særlige forsiktighetsregler ved bruk

Tunnelrestriksjoner: Kategori E

### 14.7. Bulktransport i henhold til vedlegg II i MARPOL 73/78 og IBC-regelverket

Ikke aktuelt

#### 14.8 Annen transportinformasjon

Stuing: Kategori A

Nödinstruksjonner ( EmS ):ved BRANN ( IMDG ) F-A

Nödinstruksjoner ( EmS ) ved UTSLIPP (IMDG ) S-B

Transportkategori: 2: Höyeste totale mengde per transportenhet 333 kg eller liter

## AVSNITT 15: OPPLYSNINGER OM BESTEMMELSER

### 15.1. Særlige bestemmelser/særskilt lovgivning om sikkerhet, helse og miljø for stoffet eller stoffblandingen

Ikke aktuelt.

### 15.2. Vurdering av kjemikaliesikkerhet

Vurdering og kjemikaliesikkerhetsrapport henhold til 1907/2006 Vedlegg I er ennå ikke gjort.

## AVSNITT 16: ANDRE OPPLYSNINGER

### 16a. Informasjon om hvilke endringer som er gjort av den forrige versjonen

#### Revisjoner av dokumentet

Dette er den første utgave.

### 16b. Forklaring av forkortelser i sikkerhetsdatabladet

#### Fulltekst for koder for fareklasse og kategori er nevnt i Avsnitt 3

| | |
|---|---|
| Acute Tox 4*oral* | Akutt giftighet (Kategori 4 svelging) |
| Eye Dam 1 | Irreversible øyevirkninger (Kategori 1) |
| Skin Corr 1A | Etsende (kategori 1A) |
| Skin Irrit 2 | Irriterende for huden (Kategori 2) |

#### Detaljert definisjon av farene som nevnt i punkt 2

**Skin Corr 1B**

På grunnlag av resultatene fra dyreforsøkene klassifiseres stoffet som etsende, underkategori 1B etter 1272/2008 Vedlegg I ), dvs. det forårsaker synlig nekrose gjennom epidermis og ned til dermis hos minst 1 av 3 forsøksdyr ved eksponering i mer enn 3 minutter, men ikke mer enn 1 time ved en observasjonstid på 14 dager. Typiske skader som forårsakes av etsende stoffer, er sår, blødninger og blodige sårskorper. På slutten av observasjonsperioden på 14 dager forekommer også misfarging av huden på grunn av bleking, partier med håravfall og arr

#### Forklaringer til forkortelser i avsnitt 14

| | |
|---|---|
| ADR | Europeisk avtale vedrørende internasjonal transport av farlig gods på vei |
| RID | Reglementet for internasjonal transport av farlig gods med tog |

### 16c. Kildene til de viktigste data brukt ved utarbeidingen av sikkerhetsdatabladet

#### Datakilder

Primærdata for beregning av farene har først og fremst blitt hentet fra den offisielle europeiske klassifikasjonslisten, 1272/2008 Vedlegg I, oppdatert til 2015-04-13.

Der slike oppgaver mangler,ble det i andre hånd brukt den dokumentasjonen som ligger til grunn for den offisielle klassifiseringen, f.eks. IUCLID (International Uniform Chemical Information Database). I tredje hånd ble informasjonen fra ansette internasjonale kjemikalieforetak brukt, og i fjerde fra annen tilgjengelig informasjon, f.eks. fra andre leverandørers sikkerhetsdatablader eller fra ideelle organisasjoner, der en ekspertbedømmelsen har blitt foretatt av kildens troverdighet. Hvis pålitelig informasjon ikke finnes til tross for dette, har farene blitt bedømt av ekspertise på grunnlag av kjente farer fra lignende stoffer, der prinsippene i 1907/2006 og 1272/2008 har blitt fulgt.

#### Fulltekst for forskrifter som er nevnt i dette sikkerhetsdatabladet

| | |
|---|---|
| 453/2010 | Kommisjonsforordning (EU) nr. 453/2010 av 20. mai 2010 om endring av Europaparlaments og rådsforordning (EF) nr. 1907/2009 om registrering, vurdering, godkjenning og begrensninger av kjemikalier (REACH) |
| 1272/2008 | EUROPAPARLAMENTS- OG RÅDSFORORDNING (EF) nr. 1272/2008 av 16. desember 2008 om klassifisering, merking og emballering av stoffer og blandinger, om endring og oppheving av direktiv 67/548/EØF og 1999/45/EF, og om endring av forordning (EF) nr. 1907/2006 |
| 1999/45/EG | Europaparlaments- og rådsdirektiv 1999/45/EF av 31. mai 1999 om tilnærming av medlemsstatenes lover og |

| | |
|---|---|
| | forskrifter om klassifisering, emballering og merking av farlige stoffblandinger |
| 89/391 | Europaparlaments- og rådsdirektiv 89/391/EF |
| 98/24 | Europaparlaments- og rådsdirektiv 98/24/EF |
| 1907/2006 | EUROPAPARLAMENTS- OG RÅDSFORORDNING (EF) nr. 1907/2006 av 18. desember 2006 om registrering, vurdering og godkjenning av samt begrensninger for kjemikalier (REACH), om opprettelse av et europeisk kjemikaliebyrå, om endring av direktiv 1999/45/EF og om oppheving av rådsforordning (EØF) nr. 793/93 og kommisjonsforordning (EF) nr. 1488/94 samt rådsdirektiv 76/769/EØF og kommisjonsdirektiv 91/155/EØF, 93/67/EØF, 93/105/EF og 2000/21/EF Vedlegg I |

## 16d. Metoder for å evaluere opplysningene i henhold til 1272/2008 Artikkel 9 brukt i klassifiseringen

Beregningen av farene med denne blandingen er gjort som en samveid bedømmelse med hjelp av en ekspertbedømmelse i samsvar med 1272/2008 Vedlegg I, der all tilgjengelig informasjon som kan ha betydning for å fastsette farene med blandingen veies sammen, og i samsvar med 1907/2006 Vedlegg XI.

## 16e. Lister over relevante R-setninger, faresetninger og sikkerhetssetninger

### Fulltekst for faresetninger henhold til GHS/CLP under Avsnitt 3

| | |
|---|---|
| H314 | Gir alvorlige etseskader på hud og øyne |
| H302 | Farlig ved svelging |
| H318 | Gir alvorlig øyeskade |
| H315 | Irriterer huden |

## 16f. Råd om passende opplæring for ansatte for å beskytte menneskers helse og miljøet

## Annen relevant informasjon

# E Principal component analysis: scripts

**R script for setting up the correlation matrix for principal component analysis, developed by A. Ferry, University of Lyon, <apolline.ferry@insa-lyon.fr>.**

```
#Clear the R environment.
rm(list=ls())

#R packages needed to make correlation matrix.
install.packages('corrplot', 'Hmisc', 'PerformanceAnalytics')

#Load text file with data, "file path.txt"
#sep="\t" gives that delimiter is tab,
#header=TRUE  means that your columns have name in the first rows.
data_S <- read.table("file path.txt", sep="\t", header=TRUE)

#R's integrated function cor() uses the Pearson correlation coefficient
#as default, which measures the linear dependence between two variables.
#"mcor" is the output correlation matrix containing the correlation
#coefficients between each variable and the others.
mcor <- cor(data_S,use="complete.obs")

#The function rcorr() in the Hmisc package computes
#the significance levels for all Pearson correlations.
library(Hmisc)
res=rcorr(as.matrix(data_S[,1:20]))
signif(res$r, 2)
signif(res$P,2)

#flattenCorrMatrix is custom a function, formatiing the correlation matrix
#into a table of four columns:
#"row" and "column" (variable 1 and 2 for the correlation test),
#"cor" (correlation coefficients) and "p" (p-values of the correlations).
flattenCorrMatrix <- function(cormat, pmat) {
  ut <- upper.tri(cormat)
  data.frame(
    row = rownames(cormat)[row(cormat)[ut]],
    column = rownames(cormat)[col(cormat)[ut]],
    cor  =(cormat)[ut],
    p = pmat[ut]
  )
}

B=flattenCorrMatrix(res$r, res$P)
```

```

#Draw scatterplot. Output not used in this thesis.
library(PerformanceAnalytics)
chart.Correlation(data_S, histogram=TRUE, pch=19)

#Create a correlogram, display only the "upper"
#triangular of the correlation matrix.
library(corrplot)
corrplot(mcor, type="upper", order="hclust", tl.col="black", tl.srt=45)

#Choose layout and colors.
col<- colorRampPalette(c("blue", "white", "red"))(20)
heatmap(x = mcor, col = col, symm = TRUE)

#Replace correlation coefficients by symbols according to the level
#of correlation. abbr.colnames: Column names were not abbreviated.
symnum(mcor, abbr.colnames=FALSE)
```

**R script for performing principal component analysis, developed by A. Ferry, University of Lyon, <apolline.ferry@insa-lyon.fr>.**

```r
#Clear the R environment.
rm(list=ls())

#Download Factoshiny, a web page used to practice multivariate analysis.
#Developed and maintained by Francois Husson et.al.,
#husson@agrocampus-ouest.fr
download.packages(Factoshiny)
download.packages(FactoMineR)

#Get the input file from local "file path.txt", sep="\t"  is used
#to set delimiter as tab, header=TRUE  sets  the first row as column names.
data_S <- read.table("file path.txt", sep="\t", header=TRUE)

library(Factoshiny)
ACPbis<-Factoshiny(data_S)
library(FactoMineR)

#quanti.sup used to choose quantitative columns to not include
#in the PCA. In this case, it was the columns
#"Column position", "Column number" and "Day".
#These are are then indicated in blue in output PCA plot.
#quali.sup uses the same principle, but for qualitative columns.
ACP<-PCA(data_S,quanti.sup = 1:3,quali.sup=20)

#summary() used to obtain a summary of the PCA
#(eigenvalues,coordinates, etc.).
#nbelements=Inf to obtain information about all the individuals.
summary(ACP,nbelements=Inf)

#Open a new window.
x11()

#Plot the data.
#habillage is used to choose the variable to color the data in.
plot(ACP,habillage=2)
x11()
plot(ACP,choix="var")
```

# F Complementary plots from principal component analysis

**Score and loading plots from principal component analysis, made by A. Ferry, University of Lyon, <apolline.ferry@insa-lyon.fr>.**

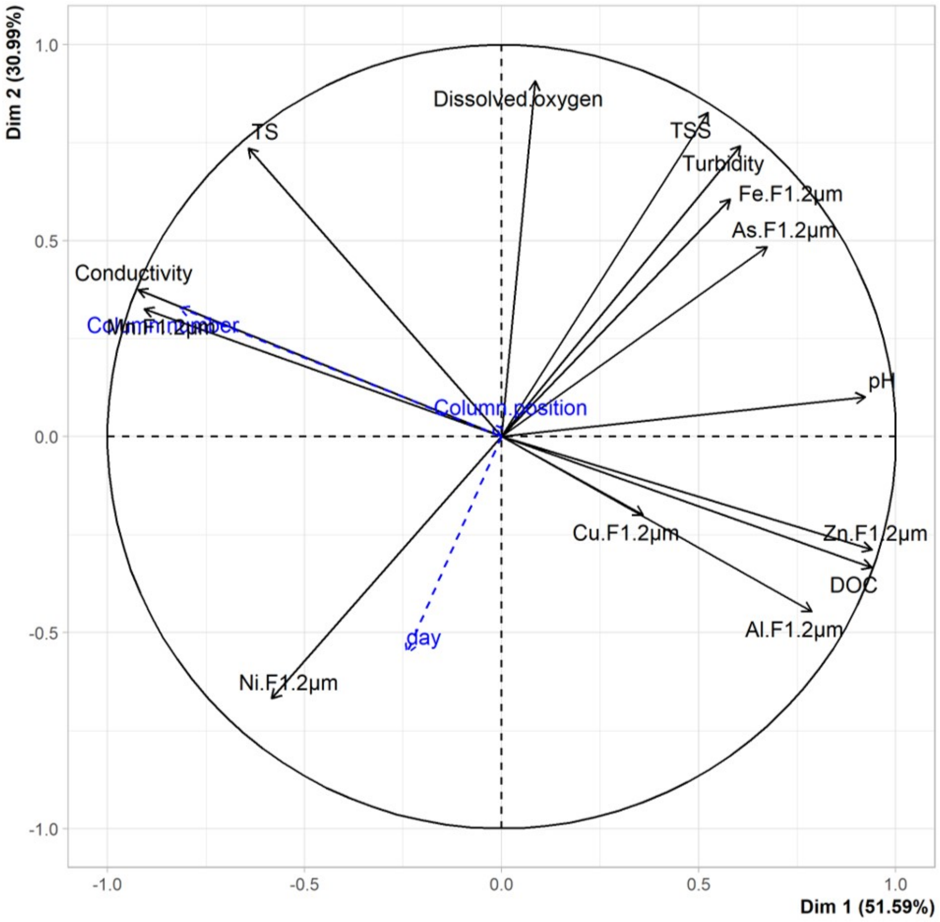


Figure 29: Loading plot from principal component analysis of general water quality parameters and 0.45 $\mu$m metal fractions in samples from the Grillstad and Smestad tunnels.

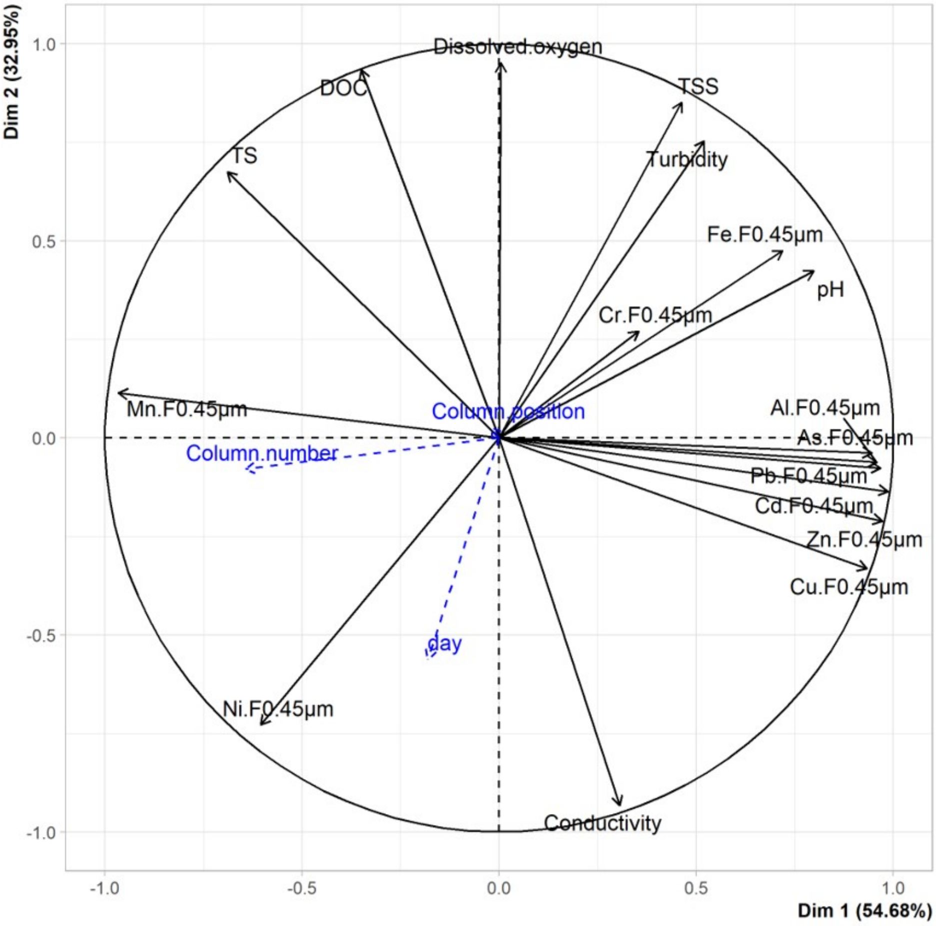


Figure 30: Loading plot from principal component analysis of general water quality parameters and 0.45 $\mu$m metal fractions in samples from the Grillstad and Smestad tunnels.

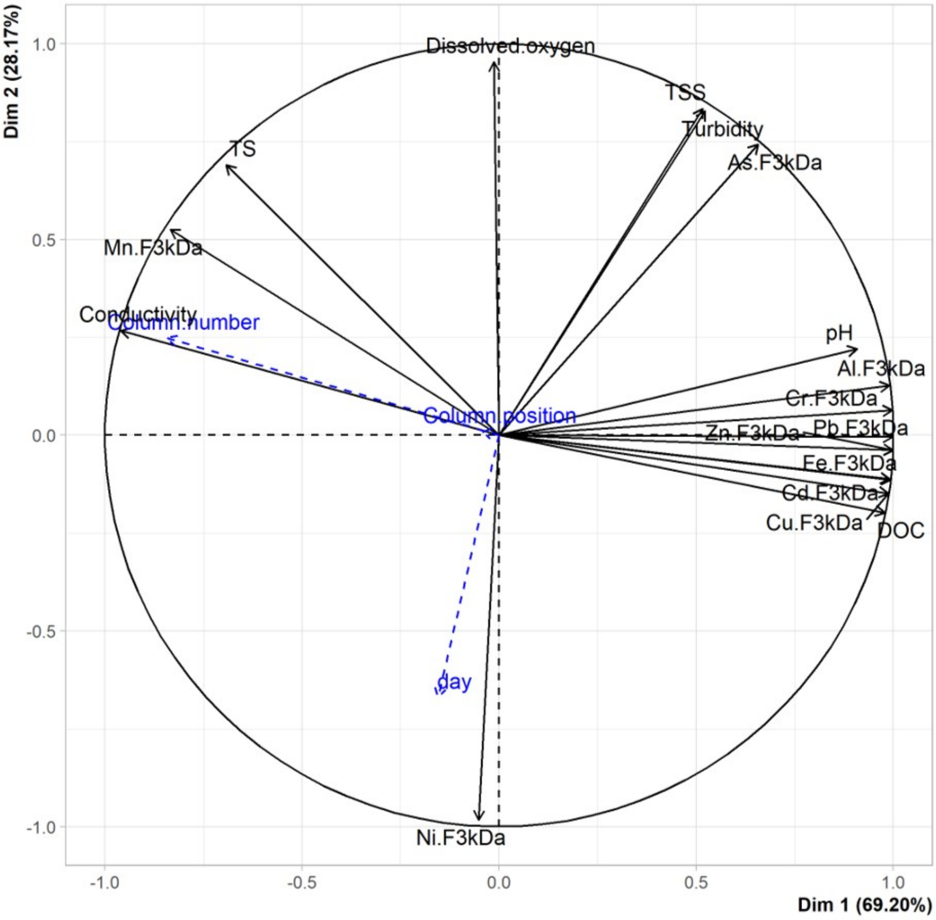


Figure 31: Loading plot from principal component analysis of general water quality parameters and 3 kDa metal fractions in samples from the Grillstad and Smestad tunnels.

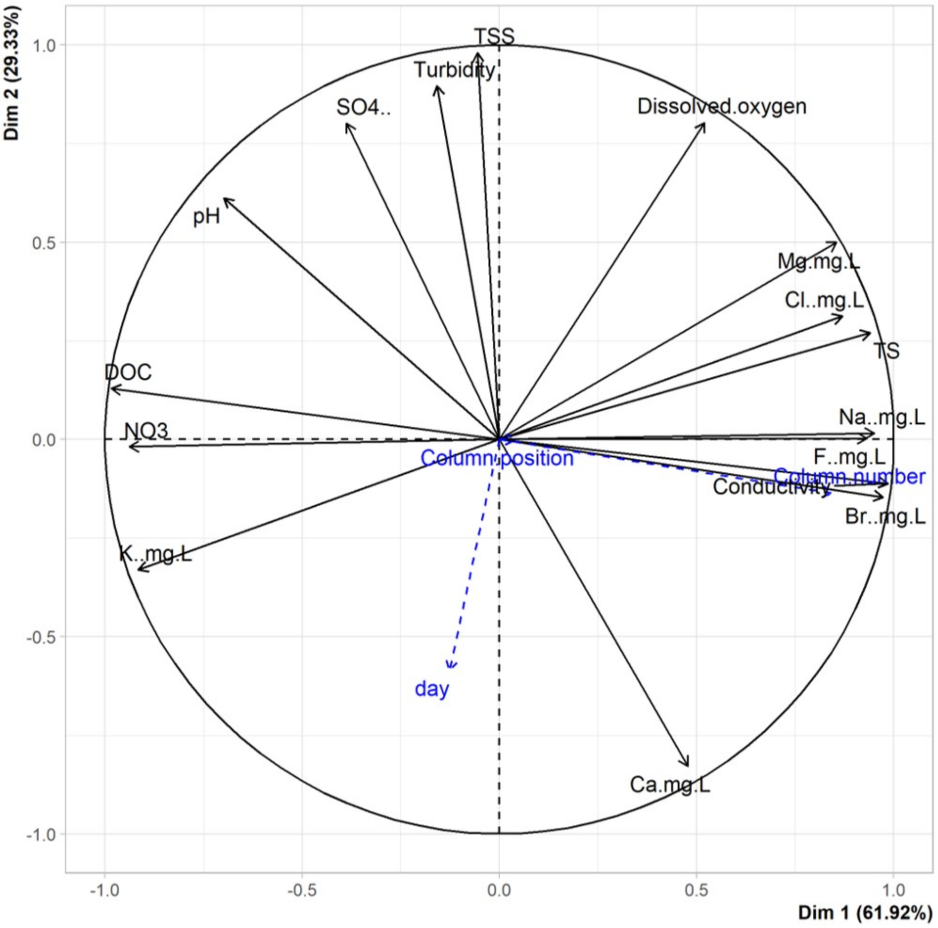


Figure 32: Loading plot from principal component analysis of general water quality parameters and ions metal fractions in samples from the Grillstad and Smestad tunnels.

Sofie Eivik Karlsen

Evolution of tunnel wash water quality during sedimentation

NTNU
Norwegian University of
Science and Technology