Contaminated landslide runout deposits in rivers – Method for estimating long-term ecological risks

Citation for the original published paper (version of record):

Goransson, G., Norrman, J., Larson, M. (2018). Contaminated landslide runout deposits in rivers – Method for estimating long-term ecological risks. Science of the Total Environment, 642: 553-566. http://dx.doi.org/10.1016/j.scitotenv.2018.06.078

(article starts on next page)

Science of the Total Environment 642 (2018) 553–566

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Contaminated landslide runout deposits in rivers – Method for estimating long-term ecological risks

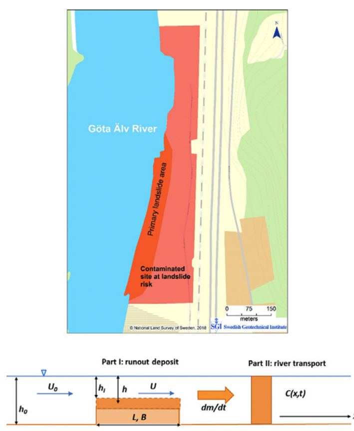
G. Göransson [a,*], J. Norrman [b], M. Larson [c]

[a] Climate Adaptation, Swedish Geotechnical Institute, SE-412 96 Gothenburg, Sweden
[b] Architecture and Civil Engineering, Chalmers University of Technology, SE-412 96 Gothenburg, Sweden
[c] Water Resources Engineering, Lund University, SE-221 00 Lund, Sweden

HIGHLIGHTS

- Analysis of effects of contaminant release from landslide runout deposits into rivers.
- A method to estimate long-term ecological risks to rivers and estuaries is developed.
- An addition to existing contaminated site and landslide risk assessment methods.
- Can be considered in integrated water management plans.

GRAPHICAL ABSTRACT



ARTICLE INFO

Article history:
Received 23 January 2018
Received in revised form 7 June 2018
Accepted 7 June 2018
Available online 14 June 2018

Editor: G. Ashantha Goonetilleke

Keywords:
Contamination
Ecological risk
Landslide
Probabilistic method
River erosion
Water quality

ABSTRACT

The potential catastrophic event of a landslide bringing contaminants to surface waters has been highlighted in public media, but there are still few scientific studies analyzing the risk of landslides with contaminated soil. The aim of this study is to present a method to estimate the risk of potential long-term ecological effects on water bodies due to contaminated soil released into a river through a landslide. The study constitutes further development of previous work focusing on the instantaneous (short-term) release of contaminants and associated effects. Risk is here defined as the probability of surface water failing to comply with environmental quality standards (EQS). The transport model formulation is kept simple enough to allow for a probabilistic analysis as a first assessment of the impact on the river water quality from a landslide runout deposit containing contaminated soil. The model is applied at a contaminated site located adjacent to the Göta Älv River that discharges into the Gothenburg estuary, in southwest Sweden. The results from the case study show that a contaminated runout deposit will likely cause contamination levels above EQSs in the near area for a long time and that it will take several years for the deposit to erode, with the greatest erosion at the beginning when water velocities are their highest above the deposit. A contaminated landslide runout deposit will thus act as a source of contamination to the downstream water system until all the contaminated deposit has been eroded away and the contaminants have been transported from the deposit to the river, and further to the river mouth – diluted but not

* Corresponding author.
E-mail addresses: gunnel.goransson@swedgeo.se, (G. Göransson), jenny.norrman@chalmers.se, (J. Norrman), magnus.larson@tvrl.lth.se (M. Larson).

https://doi.org/10.1016/j.scitotenv.2018.06.078

necessarily negligible. Therefore, it is important to prevent landslides of contaminated soil or waste, and if such events were to occur, to remove the contaminated runout deposit as soon as possible.

1. Introduction

There are few scientific studies on the combination of landslide susceptibility and contaminant spread, although landslides often affect urbanized and/or industrialized areas. Already in 1987, McCahon et al. (1987) pointed out that most studies on landslide impacts emphasize the geomorphological, social or economic aspects, and that ecological impacts are paid very little attention. McCahon et al. (1987) studied the chemical and biological effects of the Pennine peat slide into the headwaters of Langdon Beck and found that the slide caused a large increase in the concentration of suspended sediment and metals, with metal concentrations that reached acute toxic levels for aquatic organisms, but that long-term effects could not be assessed due to lack of data. Since then, little research appears to have been done on this subject. However, related cases have been reported in public media, implying an increased awareness of the potential that landslides can spread contaminants. For instance, public media has reported on a series of unusual landslides in connection with the fracking industry that have brought metal contaminated landslide sludge into the Brenot Creek and the Peace River, Canada (Business Vancouver, 2015; Policynote, 2016). Another example is from the Otago Daily Times that published an article on the Oso Landslide, Washington, US, discussing the potential that this event could cause the release of contaminants from broken sewages as well as the release of propane, household solvents, and other chemicals occurring within and beneath the landslide runout deposit (Otago Daily Times, 2017). In June 2013, the American Geophysical Union (AGU) blogosphere reported on a landslide event in Ecuador that caused a rupture of the so-called Trans-Ecuador pipeline and the leakage of oil, which led to a temporary closure of the Cocoa drinking water supply (Petley, 2013).

Reporting has also been made on the risk of landslides spreading radioactive waste products from the nuclear industry, which since World War II have been dumped in the landslide prone Ferghana Valley area that stretches across Kyrgystan, Uzbekistan, and Tajikistan (Institute of Nuclear Physics NNC RK, 2003; New Internationalist, 2005). Most of the mining facilities in this area were built on unstable slopes. The absence of a suitable location for landfill of wastes led to the dumping of radioactive wastes on floodplains adjacent to rivers (Institute of Nuclear Physics NNC RK, 2003). Underground work and other anthropogenic activities are believed to have caused many of the reported landslides in this area (Institute of Nuclear Physics NNC RK, 2003; UN-SPIDER, 2017). In 1958, a landslide in Mailuu Suu, triggered by earthquakes and heavy rains, caused the burst of a tailing that released 600,000 m^3 of radioactive waste into the nearby river (Institute of Nuclear Physics NNC RK, 2003). Catastrophe modeling of other tailings (tailing dumps and stockpiles) in the Mailuu Suu area has been carried out to analyze possible consequences. The modeling showed that if a tailing is destroyed by a landslide, the contaminated material will most likely be released into the nearby river (Birsen and Kadyrzhanov, 2003). In 2014, Torgoev and Omorov (2014) published a study on the landslide hazards of rock glaciers in the waste material from the Kumtor gold mine, Kyrgystan. They argued that such an event will most likely cause the pollution of the nearby Kumtor-Naryn River basin.

Schreier and Lavkulish (2015) studied the transport and settling of asbestos-rich serpentine sediment dispersed from a landslide area in the Sumas River Watershed in Washington State and British Columbia, and found that the transport of sediment from the landslide to the river is highest during high-flow periods (winter), and that the suspended sediment then moves through the river system in pulses of suspension, deposition, and re-suspension during several storm events. They also found that the sediment zeta-potential impacts the flocculation, coagulation, and deposition of the sediment, which can explain that particles <63 μm settled more rapidly near the landslide area than further downstream (Schreier and Lavkulish, 2015). A study by Johnston et al. (2015) found that landslides may be a significant mobilizer of Fe and Mn from the soil to the aquatic fiordland system, and that the landslides are important for the Fe and Mn cycle in the system.

Other scientific studies that may provide insights into the transport processes that are relevant for the release of contaminants from landslide runout deposits involve contaminant spread from riverbank erosion. For example, Rowan et al. (1995) studied geomorphology and pollution of the Glengonnar Water and found that bank erosion, among others, was an important process that affected the metal content in the river. Foulds et al. (2014) found that erosion in its various forms (surface erosion, rill erosion, bank erosion, and bank collapse) is an important pathway for contaminants to enter a river system. Rhoades et al. (2009) studied the release of mercury (Hg) from riverbank erosion along the South River, Waynesboro, USA. They found that fluvial bank erosion of Hg contaminated soil (from textile manufacturing) was a major source for the Hg load to the water system. Carroll and Warwick (2016) studied the mechanisms of Hg spreading (gold mining) along Carson River, Lahontan Reservoir system in Nevada, USA. They studied Hg loading from diffusion, channel pore water advective flux, bank erosion, and overbank deposition, and found that bank erosion processes were dominant for the Hg load to the river water, for both solid and dissolved Hg. Coulthard and Mackling (2003) modeled sediment and contaminant transport along the River Swale, which is affected by metal mining, in order to predict future conditions. Based on their simulations, they found that it will most likely take several thousands of years of natural erosion to remove all the contaminants from the Swale River channel and its floodplain; meanwhile the contaminants will continue to affect adjacent land and water resources.

So far, we have found no studies on risk assessment that account for contaminant release from landslide events – from hazard identification to risk analysis. To elucidate this hazard, Göransson et al. (2009) used GIS-technique to combine layers for landslide probability with layers for identified, risk-classified contaminated sites along parts of the Göta Älv River valley, in southwest Sweden. The study revealed that out of 31 identified contaminated sites, 8 sites were located in areas with unacceptable slope stability. Out of these 8 sites, 5 were assessed as posing high or very high environmental risk. Two historical landslides in the area that involved industrial sites are the Göta landslide in 1957 resulting in the failure of the Göta Sulphite factory, and the Agnesberg landslide in 1993 causing a large part of the Agnesberg industrial site to slide into the river. Some sediment samples taken from the Agnesberg landslide runout deposit show metal content. In connection with the Agnesberg landslide, the freshwater intake for the City of Gothenburg, located downstream of the slide area, registered elevated levels of turbidity resulting from the initiation slide and the following retrogressive slides at the source area (Göransson et al., 2012). Ströberg et al. (2017) suggested a method to derive a landslide susceptibility index to be used in a GIS-analysis to overlap landslide susceptibility and the location of contaminated sites along parts of the Ångermanälven River, in northern Sweden. They found that 16 of 209 potentially contaminated sites are located in areas with high or very high susceptibility for landslides.

It is possible that the effects of climate change may increase the landslide frequency in areas where precipitation is anticipated to increase. For example, IPCC states that there is a "high confidence that changes in heavy precipitation will affect landslide conditions in some regions" (Seneviratne et al., 2012). A recently performed landslide risk analysis along the Göta Älv River valley accounted for the effects of climate change. The result of the analysis showed that landslide susceptibility in cohesive fine sediments, present in the Göta Älv River valley, will increase because of excessive shear stresses during high river flows (causing erosion and steepening of the river banks) and elevated groundwater pressure in the river banks from increased precipitation (SGI, 2012; Göransson et al., 2016; Odén et al., 2017).

Göransson et al. (2014) proposed a probabilistic method to quantitatively estimate the spreading of contaminants immediately after the instantaneous release of material when landslides bring contaminated soil into a river. The probability of a landslide was combined with the probability that such an event will cause a deterioration of the water quality. The method was tested in a case study involving an area located adjacent to the Göta Älv River. The results indicated high levels of contaminants in the landslide runout deposit (the sediment), elevated contaminant concentrations in the river water above acceptable water quality criteria, as well as a significant contribution to the yearly contaminant load to the Göta Älv estuary.

The aim of the present study has been to develop a method to include potential long-term ecological risks to water bodies. The proposed method for estimating the long-term ecological risk includes the erosion of contaminated material from the runout deposit and the downstream transport and spreading in the river. The results provide an important complement to environmental risk assessments for contaminated sites and/or methods for landslide risk assessment. The method is illustrated by the same case study site that was used in the previous work, further described in this paper.

The ecological risks are investigated in two scenarios: scenario (A) considers the time immediately after the instantaneous release until dredging takes place to free the fairway (a characteristic time scale of weeks); scenario (B) considers the time after the instantaneous release until the contaminated landslide deposit has been eroded away by the flowing water (time scale of several months or years). Some of the questions investigated in the study related to the contaminant release from the deposit are:

- How long will the concentrations stay above a certain critical level?
- How large is the contribution compared to the background load?

2. Materials and method

2.1. Definition of risk

Risk can be defined in somewhat different ways, e.g., as "the chance, within a time frame, of an adverse event with specific consequences" (Burgman, 2005), as a "concept that denotes a potential negative impact on an asset" (Swartjes, 2011), or as the "effect of uncertainty on objectives" (ISO, 2009). Risk is often expressed in terms of the combination of the consequences (or effects) of an event and the associated likelihood (or probability) (e.g. ISO, 2009; UNISDR, 2009; Swartjes, 2011; IPCC, 2014), but variations exist (see e.g. Aven and Renn, 2009). Different disciplines also use the same terms in slightly different ways: risk assessment is sometimes equated to risk analysis (e.g. Swartjes, 2011); risk assessment can be seen as a part of risk analysis (e.g. Covello and Merkhofer, 1993; NRC, 1996); or risk analysis can be a part of risk assessment (e.g. Aven, 2003; ISO, 2009; SafeLand, 2011). The two former approaches typically relate to environmental (ecological and human health) risk assessment, whereas the latter relates to more technological risks. However, common features for establishing whether unacceptable risks exist are: problem definition (scope definition, establishing context), hazard identification (assessment), exposure assessment (sometimes combined with hazard assessment), and risk characterization (risk evaluation, risk estimation). Since there are variations in the exact definition of risk and associated concepts, it is a good idea to provide definitions for any given situation or analysis.

In the proposed method, the risk assessment (or risk analysis depending on discipline) is simplified in such way that it focuses on one specific hazard (landslide with contaminated soil) and specific endpoints (the surface water ecosystem at different locations and for different periods of time). The method aligns to the commonly used expression of risk as a combination of the consequences of a specific undesirable event and the probability that it will happen. The undesirable event is here specified as *surface waters of the river and estuary that fail to comply with environmental quality standards (EQS[1]) due to a landslide with contaminated soil*. The consequences of this event are not further quantified, but since EQSs are based on effects (e.g., biological) and responses (e.g., the amount affected), they indirectly express something about potential negative consequences of the event of exceeding those levels. The proposed method instead focuses on estimating the probability of the specified undesirable event. The information from the analysis is intended to provide support for decision-makers managing sites that are contaminated and simultaneously prone to landslide within a river basin.

For the purpose of estimating the probability of the defined event it is useful to look at the risk as a chain of events, from the stressor (the contaminated soil) to the endpoint (the ecosystem in the river and in the estuary), at which a negative effect is assumed to occur when EQSs are exceeded. Here, the probability of the event of exceeding relevant EQSs is associated with the probabilities of four preceding events:

Event 1: The soil is contaminated.

Event 2: The contaminated soil slides into the river.

Event 3: Contaminants in the landslide runout deposit are released.

Event 4: The released contaminants in water reach concentrations that exceed relevant EQSs.

If the probabilities of Events 1 and 3 are assumed to be 1.0 for contaminated areas and for the landslide runout deposit in such areas, the probability of failing to comply with EQSs in the river water and the estuary can be estimated by the combination of Event 2 (the probability of a landslide) and Event 4 (the conditional probability of exceeding a certain compliance level in case of a landslide).

As a clarification to Event 4, the present study focuses on the period after the instantaneous impulse release when there is a constant transport of contaminants from the runout deposit in the river due to erosion of the deposit. The release of contaminants through erosion is expected to continue until dredging occurs and the contaminated part of the landslide deposit is removed (Scenario A), or until there is no contaminated sediment left in the runout deposit (Scenario B).

2.2. Method for estimating the risk

The method proposed here, and applied in the case study, includes the following steps:

1. Define failure criteria.
2. Calculate the probability of landslide occurrence for the specific site (P_L).
3. Conceptualize and mathematically describe an analytical model for long-term release of contaminants from the landslide runout deposit.
4. Set parameters and estimate their uncertainty distributions in the analytical model.
5. Run Monte-Carlo simulations to derive an estimate of the probability to exceed defined compliance levels.
6. Calculate the resulting probability of failure (P_f), i.e., to exceed failure criteria.

[1] "An Environmental Quality Standard is a value, generally defined by regulation, which specifies the maximum permissible concentration of a potentially hazardous chemical in an environmental sample", http://www.gesamp.org/work-programme/eqs.

7. Perform a sensitivity analysis to obtain information about which parameters contribute most to the uncertainty of the result, and possibilities to reduce the uncertainty.

All steps are described below and implemented in the case study application section.

2.2.1. Step 1. Failure criteria

The prolonged impact phase can be considered as a long-term exposure for the organisms that will continue until contaminants from the runout deposit are no longer available. The suggestion is therefore to use chronic toxicity levels (i.e., a concentration) for fresh and/or marine aquatic life. In addition, we are also interested in the contaminant load to the river estuary; particle settlement and dilution causes the concentration to be very low, yet the contribution is a supplement to the annual load and may thus contribute to negative consequences for the ecosystem.

Failures as indicators for potential harm to the ecosystem in relation to the long-term effects from the contaminated runout deposit, are broadly defined as follows:

- Exceeding a critical concentration (EQS) during a specified time. For scenario A, this time is limited to weeks (i.e., until dredging), whereas for scenario B, the time is months to years.
- Exceeding a critical total load to the estuary. For scenario A, the contribution to this load is limited to a time of weeks (i.e., until dredging), whereas for scenario B, the contribution to this load continues for months to years.

2.2.2. Step 2. Landslide probability

The landslide probability is the probability that a hazard (landslide) will occur at a specific site. A landslide is the downward mass movement of slope-forming materials, such as soil, rock, artificial fills, or a combination of these, resulting from nature's pursuit of equilibrium (USGS, 2004). A landslide can be classified depending on the type of soil (rock, debris, soil) and the type of movement (for example, falling, sliding, and flowing). One of the most well-known classification systems for categorizing different types of mass movements is that by Varnes from 1978 (see examples in USGS, 2004). The type of landslide that is of relevance for the Göta Älv River valley is classified as a *rotational slide* in cohesive soils, which means that a coherent mass of soil is rotating around an axis of momentum perpendicular to the watercourse (Fig. 1). This movement is initially quite rapid and in situations where high sensitive clays are involved such slides can rapidly become very large.

Slope stability is governed by the equilibrium between the driving forces (i.e., external, permanent, and temporary loads, high groundwater pressure in the slope, the weight of the soil, and shear stresses at the toe of the slope from water currents) and the resisting forces (i.e., soil strength, undrained and drained shear strength, water pressure (e.g., river) at the toe of the slope, and negative pore water pressure in silty clay); if the driving forces exceed the resisting forces the slope fails.

The probability that a slope will fail and generate a mass movement is mainly a function of physical properties (slope gradient and elevation, geotechnical properties, vegetation cover, etc.) and triggering factors (rains, floods, earthquakes, land-use and other anthropogenic activities etc.) (Dai et al., 2002; Motamedi, 2013). In the review paper by Dai et al. (2002), frequency analysis, heuristic approaches, deterministic approaches, and statistical models are mentioned as methods for assessing landslide probability. In our study, we use an analytical method to analyze the probability for slope failure presented by Alén (1998) and Berggren et al. (2011). In short, the method starts with the development of an analytical solution for the probability of a landslide, based on a traditional model for slope stability calculation, incorporating parameter uncertainty. Then, the time aspects are considered by specifying parameters with variability in time. The probability for a longer time period is then obtained by modeling the slope as a sequential system with each year as a component. In this study, a time period of 50 years is considered, and the change that affects the probability of a landslide over time is attributed to river erosion (changes in slope geometry). (See Göransson et al. (2014) for a detailed description, including calculation formulas.)

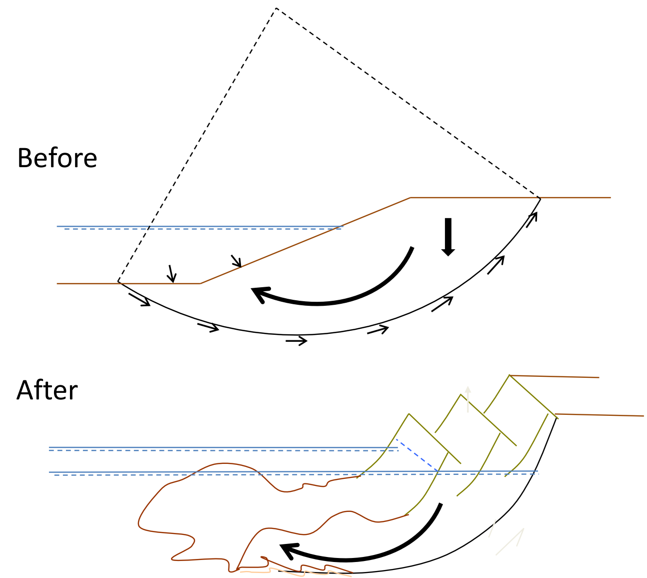


Fig. 1. Illustration of a rotational landslide perpendicular to a watercourse: a coherent soil mass is moving around an imagined axis of rotation. This type of landslide occurs mainly in fine cohesive soils and is the type of slide that is of relevance for the Göta Älv River valley.

2.2.3. Step 3. Model for the contaminant release

The conceptualization involves a homogeneous soil mass sliding by a rotational movement. The mean concentration of the contaminated part of the landslide runout deposit represents an average level of contaminants that are available for redistribution and further spread.

During the movement, contaminated material may be released to surface waters, such as lakes and rivers, and then transported in the water until conditions become favorable for deposition. In a river, the water velocities and associated turbulence and shear stresses are an order of magnitude larger than in a lake, facilitating transport and spreading of contaminated material to a much larger degree. The material from the landslide is released and transported in the water through different mechanisms operating at a wide range of scales. In the initial phase, immediately after the landslide has occurred, suspended material is directly transported by the currents in the receiving water. However, other material is deposited on the bed and gradually transported away; eventually the river returns to the morphological conditions present before the landslide when typically, dynamic equilibrium prevails. The transport away (erosion) from such a runout deposit may take place over a long period of time, depending on the transport conditions in the river and the material deposited. If sufficiently coarse material is deposited on the bed, the flow may not be able to erode all the material and may result in permanent changes to the bathymetry. This is often the case in lakes, where the transporting capacity related to the water movement is limited.

The focus herein is to develop a model of the contaminant transport from a landslide runout deposit in a river, including mobilization of material from the deposit and subsequent transport downstream in the river, not including the transport in connection with the initial, instantaneous release of material in the water when the slide occurs.

2.2.4. Steps 4 and 5. Probabilistic analysis of contaminant release

The estimation of the probability to exceed the defined failure criteria given a landslide (Step 5) is done by propagating the uncertainty

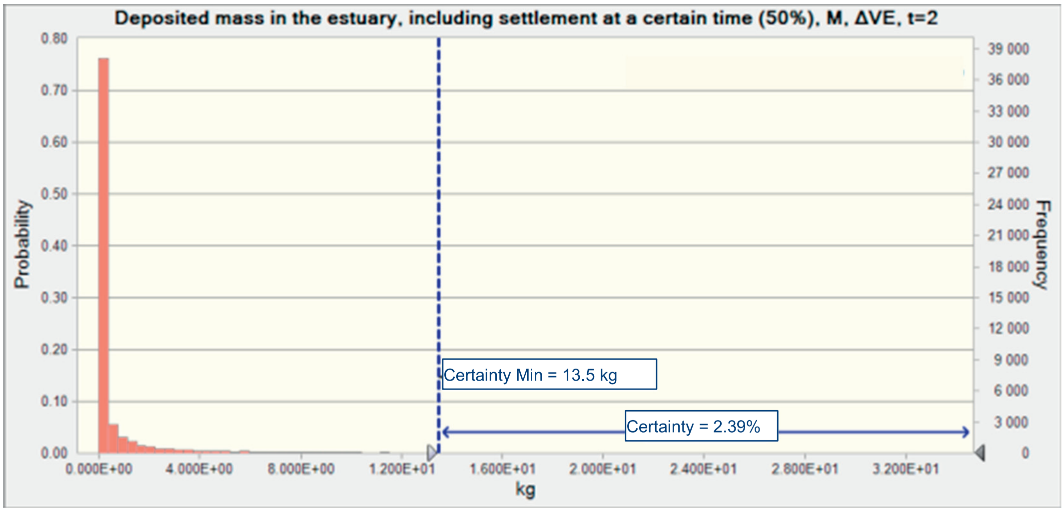


Fig. 2. Example of output distribution, showing the distribution for deposited mass of Hg in the estuary after 0.5 years, $t = 2$ (given landslide). The probability of the mass to be larger than 50% of the yearly annual background load of Hg (27 kg) is shown as ~2.4%.

associated with the model input parameters by means of Monte-Carlo simulations. Here, Crystal Ball, which is a commercial add-in to Excel, was used for the simulations and 50,000 iterations with random sampling from the uncertainty distributions were run. The results of the sought quantities are uncertainty distributions from which the probabilities to exceed the defined criteria can be derived (see the example in Fig. 2).

2.2.5. Step 6. Probability of failure

The probability of failure describes the probability that a specified negative event (failure) will occur. According to Section 2.2, the probability of failure is calculated as:

$$P_f = P_{Event\ 1} \times P_{Event\ 2} \times P_{Event\ 3} \times P_{Event\ 4} = 1 \times P_L \times 1 \times P[ZNfc \mid L] \quad (1)$$

where P_f is the probability for failure, P_L is the probability of landslide, and $P[Z > fc \mid L]$ is the probability (P) that a runout deposit will cause a contamination (Z) above a certain failure criterion (fc) given that a landslide (L) occurs.

2.2.6. Step 7. Sensitivity analysis

A sensitivity analysis provides information about which of the input variables and parameters have the largest influences on the outcome. The software Crystal Ball computes rank correlation coefficients between every assumption (uncertain input variable) and each forecast (output variables) during the Monte-Carlo simulations that show the relationship between assumptions and forecasts. To investigate the impact of input parameters that are not uncertain, a sensitivity index can be calculated (Burgman, 2005).

2.3. Case study application

2.3.1. The study site

The case study site is the former glass manufacturing plant of Surte (Surte 2:38), located adjacent to the Göta Älv River, 18 km north of Gothenburg city, in southwestern Sweden (Fig. 3). The plant was in operation between the years 1862 and 1978, leaving remnants from 115 years of activity in the form of polluted soil and groundwater. The part of the facility that is considered in this study covers an area of 30,000 m^2. Environmental investigations conducted by the municipality estimated that the soil contains about 260 t of lead, 300 t of copper, 260 t of zinc, 2.5 t of mercury, 4 t of arsenic, and a large amount of oil residues (Ale Municipality, 2014). Contaminants were continuously leaking into the Göta Älv River, and the soil slope stability was considered very low, which meant that parts of the contaminated area were at risk of sliding into the river. The highest contaminant concentrations were found closest to the river. Overall, the contamination of the site was considered to constitute a significantly increased ecological risk to the ecosystem in the river and to the raw water intake for the Gothenburg drinking water supply that is located approximately 8.4 km downstream the site (Fig. 3).

The Göta Älv River flows through fine sediments deposited in a rift valley after the latest glacial period. The sediments are dominated by clay and silt with layers of sand. At some places these mostly marine sediments reach >100 m in thickness above the bedrock. The Göta Älv River valley has the highest landslide frequency in Sweden and many of the slides have occurred in clayey soil with very high sensitivity, making the clay behave like a liquid (so called quick clay) when it starts to move. This means that landslides may potentially cover large extensions, with large consequences for the environment.

The river is a fairway for the transportation of goods to and from the industries located along it and around Lake Vänern. The river is a national priority in terms of the protection of reproduction areas for eel and salmon, and it supplies about 700,000 inhabitants in the area with drinking water. The water quality is considered good with low background concentrations of metals and organic contaminants (Göta Älvs vattenvårdsförbund, 2015a, 2015b).

In this study, we focus on the part of the former glass manufacturing site that has low slope stability and that was previously assessed to have high landslide probability (SGI, 2012). The study considers effects primarily in the near field and the recipient in the far field, the estuary. The particle bound transport of contaminants is studied and possible implications for the water quality in the near and far field are investigated for the two scenarios: (A) dredging after two weeks and (B) no dredging. The contaminants of interest in this study are lead (Pb) and mercury (Hg).

2.3.2. Defining failure

To date, there are no national environmental water quality standards for Pb and Hg. The Canadian Council of Ministers of the Environment's water quality guidelines for the protection of aquatic life in fresh water and marine water (CCME, 2016) are thus used as failure criteria. CCME does not separate between acute toxicity and chronic toxicity. Three different types of failures (I, II, III) are defined for Scenario A (dredging after 2 weeks) and Scenario B (no dredging), see Table 1. For Scenario B, we have chosen to investigate three different time periods based on calculations of the estimated time to erode the landslide deposit as described in Section 3.1.

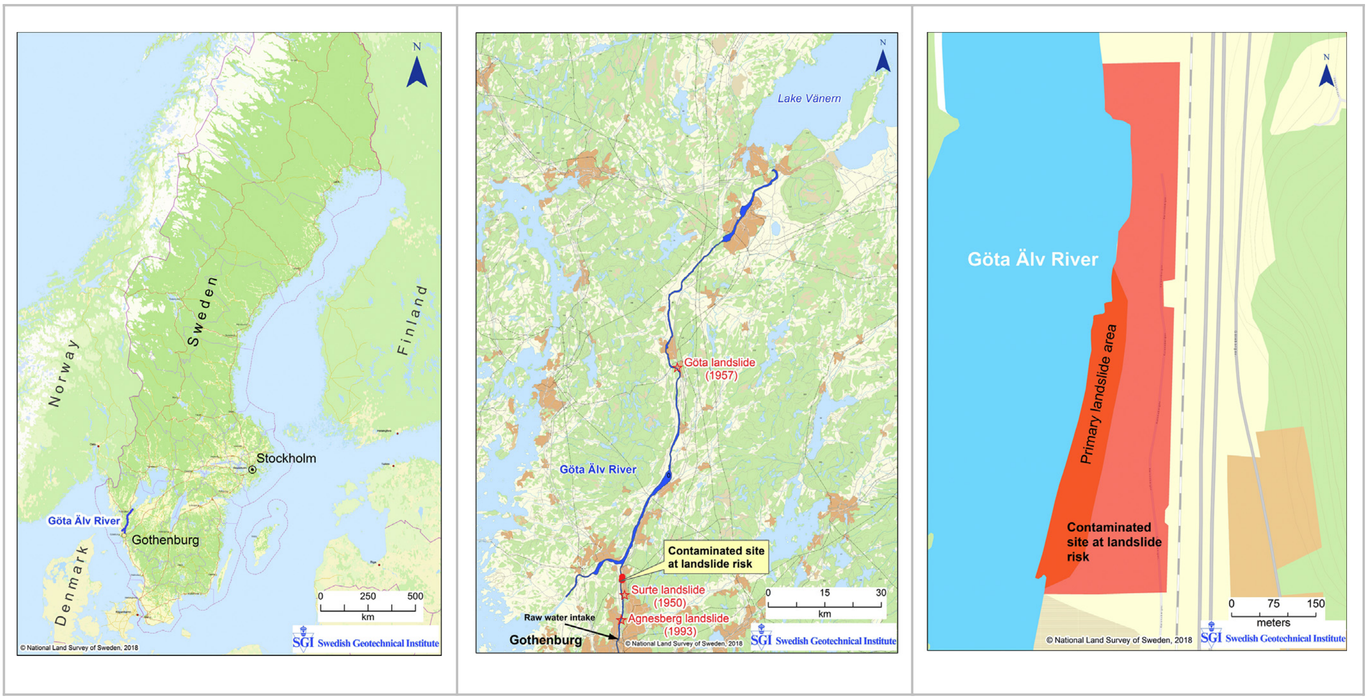


Fig. 3. Left: Map of Sweden with the Göta Älv River. Middle: The Göta Älv River and the locations: Surte 2:38, three historical landslides, and the raw water intake. Right: The Göta Älv River and the location of the study site Surte 2:28.

Table 1
The defined types of failures for the case study site, the exceedance (failure) criteria, the distance from the landslide at which failure is defined. The scenarios and associated time periods, and the mathematical expression of the defined failures.

Type of failure		Failure criteria (f_c)		Distance	Scenario	Time	Mathematical expression
No.	Description	Pb	Hg	[m]	A/B	t	
I.	Exceeding the toxicity concentration levels for fish and other living organisms in fresh water ($c_{Aq.\ fresh\ w}$) for Pb or Hg in the river water just downstream of the runout deposit during time period t.	$c_{Aq.\ fresh\ w,\ Pb} = 1$ μg/l	$c_{Aq.\ fresh\ w,\ Hg} = 0.026$ μg/l	0	A: dredging after 2 weeks B: no dredging	$t_1 = 2$ w. $t_2 = 0.5$ yrs. $t_3 = 1$ year $t_4 = 2$ yrs.	$c_0 > c_{Aq.\ fresh\ w,\ t=1}$ $c_0 > c_{Aq.\ fresh\ w,\ t=2,\ 3,\ 4}$
II.	Exceeding toxicity concentration levels for fish and other living organisms in marine water ($c_{Aq.\ marine\ w}$) for Pb or Hg in the water at the point where the river enters the estuary[a] during time period t.	$c_{Aq.\ marine\ w,Pb} = 8.1$ μg/l	$c_{Aq.\ marine\ w,Hg} = 0.016$ μg/l	18,000	A: dredging after 2 weeks B: no dredging	$t_1 = 2$ w. $t_2 = 0.5$ yrs. $t_3 = 1$ year $t_4 = 2$ yrs.	$c_0 > c_{Aq.\ marine\ w,\ t=1}$ $c_0 > c_{Aq.\ marine\ w,\ t=2,\ 3,\ 4}$
III.	Exceeding 1%, 10% or 50% of the total annual background load (W_{bg}) of Pb or Hg to the estuary[a] during time period t.	$W_{bg,\ Pb} =$ 9000 kg/yr $1\%W_{bg,Pb} =$ 90 kg/yr $10\%W_{bg,Pb} =$ 900 kg/yr $50\%W_{bg,Pb} =$ 4500 kg/yr	$W_{bg,\ Hg} = 27$ kg/yr $1\%W_{bg,Hg} =$ 0.27 kg/yr $10\%W_{bg,Hg} =$ 2.7 kg/yr $50\%W_{bg,Hg} =$ 13.5 kg/yr	>18,000	A: dredging after 2 weeks B: no dredging	$t_1 = 2$ w. $t_2 = 0.5$ yrs. $t_3 = 1$ year $t_4 = 2$ yrs.[b]	$W_{t=1} > 1\%W_{bg}$ $W_{t=1} > 10\% W_{bg}$ $W_{t=1} > 50\% W_{bg}$ $W_{t=2,\ 3,\ 4} > 1\% W_{bg}$ $W_{t=2,\ 3,\ 4} > 10\%W_{bg}$ $W_{t=2,\ 3,\ 4} > 50\%W_{bg}$

[a] The estuary is defined as the river mouth at the Bridge Älvsborgsbron.
[b] For the time period of 2 years, the total annual background load is defined as 2 × the annual background load.

2.3.3. Landslide probability

Calculation of landslide probability (P_L) was previously done by Göransson et al. (2014) showing a probability for the site of 0.3% for a 50-year time period, considering changes in geometry due to ongoing river bank and bed erosion.

2.3.4. Probability of contaminant release

The input variables in the model described in Section 3.1 are either assumed to be fixed values or represented by an uncertainty distribution (Table 2). Fixed parameters are the shape of the landslide runout deposit as well as the water depth of the river. The distance to the estuary and the compliance levels are also defined as constant values. There are five variables considered to be uncertain: the river flow rate, the mean concentration (of Pb and Hg, respectively) in the landslide deposit, the erodibility coefficient, the settling velocity, and the drag coefficient. In addition, a correlation is defined between the settling velocity and the erodibility coefficient (+0.5) since both these variables are depending on the material characteristics of the runout deposit.

3. Theory

3.1. Material transport from a landslide runout deposit

The transport of material from the runout deposit is divided into two parts (Fig. 4): (1) the erosion of material from the deposit (I: near field), and (2) the downstream transport and spreading in the river (II: far field). For the former part, two different descriptions were initially employed depending on the dominant transport mechanism; both are based on the sediment continuity equation, but the transport equations used are either for cohesive transport (Sanford and Maa, 2001) or for bed-load transport (Meyer-Peter and Mueller, 1948). In the far field, the Advection-Dispersion Equation (ADE) was first employed in the

Table 2
Input variables and chosen probability distributions for the uncertain input variables.

Input variable		Unit	Distribution	Most likely/mean	Min/0.5/0.1	Max/0.95/0.99	Stdv	Loc	Comments
Deterministic input parameters									
Deposit width (cross river)	B	m		1300					
Deposit length (cross river)	L	m		300					
Deposit thickness		m		2					h_0–h_I
Water depth, average	h_0			5					
Initial water depth over the deposit	h_I	m		3					
Pb guideline value, near field	$C_{fc,Pb}$	mg/l		0.001					CCME, 2016
Pb guideline value, far field	$C_{fc,Pb}$	mg/l		0.0081					CCME, 2016
Hg guideline value, near field	$C_{fc,Hg}$	mg/l		0.000026					CCME, 2016
Hg guideline value, far field	$C_{fc,Hg}$	mg/l		0.000016					CCME, 2016
Distance from runout deposit to the river mouth	X_R	m		18,000					
Density water	ρ_w	kg/m^3		1000					
Uncertain input variables									
Mean concentration in runout deposit (Pb)	$C_{soil,Pb}$	mg/kg ds	Lognormal	6580		17,710		771	The uncertainty around the mean is assumed lognormal distributed. The location is set to 771 (median of the sample), and the 95% to a calculated UCLM95, in Göransson et al. (2014).
Mean concentration in runout deposit (Hg)	$C_{soil,Hg}$	mg/kg ds	Lognormal	39.21		105.28		4	See above.
Density landslide runout deposit	ρ_s	kg/m^3	Triangular	1800	1700	1900			In Göransson et al. (2014)
Mean river flow	Q	m^3/s	Lognormal	161.2	98.63	255.13	28.57		Based on daily average
Settling velocity	w	m/s	Lognormal	0.001	0.0002	0.002			Assumed to be mainly clay (0.0006 mm is the particle size limit between clay and fine clay, 0.002 = fine silt). Contaminants are mainly bound to very fine particles.
Erodibility coefficient	K_T	kg/m^2 s	Lognormal	0.0000266	1.00E-06	1.00E-04			A value of 2.60*10^-5 (kg/m^2 s) was used in a previous study on Göta Älv River, river section Bohus-Göteborg.
Drag coefficient	C_D	–	Normal	0.00900	0.00600	0.01200			Can vary a lot. We have used $6 * 10^{-3}$ and $1.2 * 10^{-2}$.
Chosen points of observation									
Time/duration (t)	t0	s		0					0
	t1	s		120,960					2 week
	t2	s		15,768,000					0.5 years
	t3	s		31,536,000					1 year
	t4	s		63,072,000					2 years
Eroded portion of the deposit left ($\hat{V}$v)	V1	–		0.75					0.5 m of 2 m thickness eroded
	V2	–		0.5					1 m of 2 m thickness eroded
	V3	–		0.05					1.9 m of 2 m thickness eroded

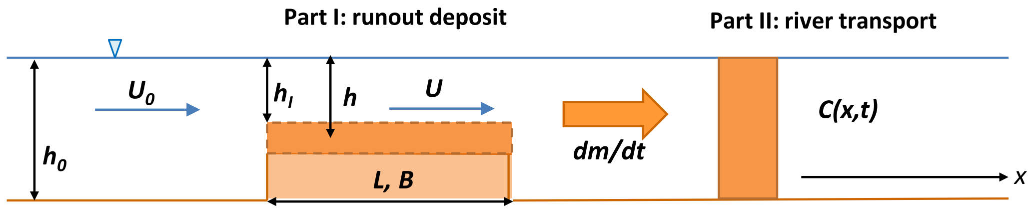


Fig. 4. Definition sketch for contaminant transport from a runout deposit in connection with a landslide to a river including the near (I) and far field (II). Explanations of the variables are given in the text.

modeling, including advection, dispersion, and sedimentation. After further investigation of the dominant processes and relevant time scales, a schematized approach was instead taken to describe the far field, where the downstream river stretch responds relatively quickly to the sediment release from the deposit. Such a schematization implies that the complete ADE in most cases is not needed, but only sedimentation should be included. For the near field, we decided to proceed with the cohesive transport formula, and the Meyer-Peter and Mueller formula was not used since it is applicable for coarser material moving along the bed, which is not of interest in the present investigation.

The runout deposit is assumed to have a rectangular shape with length (L) along the river, spanning the entire width (B). Since the water depth over the deposit (h_I and h, where h_I is the initial water depth and h is the water depth after a certain time of erosion) is smaller than the depth along the unaffected river (h_0), the velocity will be larger over the deposit, implying a greater capacity for sediment transport.

It is important to note that the model is not employed in cases where the landslide is large enough to dam the river (i.e., most of the cross-sectional areas is blocked), implying that h_I attains a small value. Such an event may involve other mechanisms that are not included here, for example complex flow conditions and dam burst effects.

Below is the summary of the equations used to calculate the release of pollutants from a runout deposit created by a landslide into a river, focusing on the impact over longer time periods for the two scenarios described above. The details of the derivation of the governing equations are provided in the Appendix together with the main assumptions underlying these equations. The concentrations and other calculated properties related to the transport of material from the deposit are based on the mass of sediment released from the deposit. Thus, the assumption is made that specific pollutants (here, metals) are closely connected to this sediment mass in the water body.

In the near field, a sediment volume conservation equation for the deposit, combined with a sediment transport equation for cohesive material following Sanford and Maa (2001), yield the following evolution equation for the deposit:

$$\frac{dh}{dt} = C_D K_T \frac{\rho}{\rho_s} U_o^2 \left(\frac{U^2}{U_o^2} - 1 \right) \tag{2}$$

where h is the water depth over the deposit, t time, C_D a drag coefficient, K_T a sediment transport coefficient, ρ and ρ_s density of water and sediment, respectively, and U and U_o the mean water velocity over the deposit and the undisturbed river bed, respectively (see Fig. 4). Introducing the continuity equation for water flow ($U = U_o h_o / h$), Eq. (2) may be written:

$$\frac{dh}{dt} = C_D K_T \frac{\rho}{\rho_s} U_o^2 \left(\frac{h_o^2}{h^2} - 1 \right) \tag{3}$$

This differential equation can be solved analytically with the initial conditions that $h = h_I$ when $t = 0$, and the solution is obtained in implicit form as:

$$h_o \ \text{arctanh}\left(\frac{h_o (h - h_I)}{h_o^2 - h h_I} \right) - h + h_I = K_T C_D \frac{\rho}{\rho_s} U_o^2 t \tag{4}$$

In the initial phase of deposit erosion, Eq. (4) may be simplified by using a Taylor series expansion to,

$$h = \frac{1}{2}(h_I - \lambda U_o t) + \sqrt{\frac{(h_I - \lambda U_o t)^2}{4} + \frac{\lambda U_o t h_o^2}{h_I}} \tag{5}$$

where the coefficient λ was introduced for convenience, defined as:

$$\lambda = K_T C_D \frac{\rho}{\rho_s} U_o \tag{6}$$

The mass transport from the deposit may be expressed as $\dot{m} = \rho_s BLU_o((h_o/h)^2 - 1)$, where $\dot{m} = dm/dt$. Immediately downstream the deposit the concentration in the river is obtained as the mass transport from the deposit divided by the flow rate ($c_o = \dot{m}/(Bh_oU_o)$), yielding $c_o = \rho_s L \lambda ((h_o/h)^2 - 1)/h_o$. Thus, the maximum concentration in the near field will always occur at $t = 0$ for which $h = h_I$, yielding $c_{\max} = \rho_s L \lambda ((h_o/h_I)^2 - 1)/h_o$.

The duration when the concentration in the near field exceeds a certain level c_L may be calculated from,

$$t_L = \frac{1}{\lambda U_o} \left(\text{h}_o \ \text{arctanh}\left(\frac{h_o (h_L - h_I)}{h_o^2 - h_L h_I} \right) - h_L + h_I \right) \tag{7}$$

where $h_L = h_o / \sqrt{1 + (c_L h_o)/(\lambda \rho_s L)}$ and $c_L \le c_{max}$. The time it takes to erode a certain portion of the deposit may be estimated from,

$$t_V = \frac{1}{\lambda U_o} \left(\text{h}_o \ \text{arctanh}\left(\frac{h_o (h_V - h_I)}{h_o^2 - h_V h_I} \right) - h_V + h_I \right) \tag{8}$$

where $h_V = h_o - \kappa(h_o - h_I)$ and κ is an input parameter denoting the ratio of the deposit remaining ($0 \le \kappa \le 1$); conversely, $\kappa = (h_o - h_V)/(h_o - h_I)$. The eroded deposit volume at time t_V is given by $\Delta V = (h_V - h_I)BL$, which also can be expressed as $\Delta V = (1 - \kappa)V$, where V is the initial deposit volume given by $V = (h_o - h_I)BL$. In order to calculate the eroded volume, the depth at the specific time must be determined either by Eq. (5) or (4). The former one is approximate, but explicit and does not require a numerical approach.

In the far field, if it is assumed that the river system responds quickly to the mass released from the deposit in relation to changes in the release rate, at least over the river stretch that extends from the deposit to where the material eventually settles (e.g., estuary), the concentration in the river stretch will be identical to the concentration at the deposit (no sediment settling). Sediment deposition on the river bed may be included by formulating a sediment balance equation according to,

$$\frac{dc}{dx} = -\frac{w}{U_o h_o} c \tag{9}$$

where w is the fall speed. This equation has the solution:

$$c = c_o \exp\left(-\frac{w}{U_o}\frac{x}{h_o}\right) \tag{10}$$

Assuming that the distance from the deposit to the estuary is x_R, the concentration at the inflow point to the estuary (c_E) is,

$$c_E = c_o \exp\left(-\frac{w}{U_o}\frac{x_R}{h_o}\right) \tag{11}$$

For $w = 0$, $c_E = c_o$, and the concentration at the inflow to the estuary is the same as the concentration at the deposit.

Neglecting sediment settling, the equations valid in the near field may also be employed to calculate the concentration at the downstream end of the river stretch, where the river water is discharged into the estuary. This yields a conservative estimate of the sediment transport to the estuary, since all sediment that is released from the deposit will settle in the estuary and no material ends up in the river stretch. If sedimentation is included, Eq. (11) should be employed instead. The maximum value will always occur for $t = 0$, as before.

The volume (or mass) of sediment that ends up in the estuary after a certain time corresponds to the erosion of the deposit, possibly corrected for the settling that occurs along the river stretch downstream of the deposit. The eroded volume (ΔV_E; what will end up in the estuary) may be calculated from:

$$\Delta V_E = (1-\kappa)(h_o - h_l)BL \exp\left(-\frac{w}{U_o}\frac{x_R}{h_o}\right) \tag{12}$$

To calculate the duration a certain concentration is exceeded at the inflow point to the estuary, given by $x = x_R$, the equations for the near field may be employed when $w = 0$. If $w > 0$, then a correction must be made for the sedimentation using Eq. (11) to determine the concentration at the deposit (c_{LE}) that would generate c_L at the estuary according to $c_{LE} = c_L \exp((wx_R)/(U_o h_o))$. The concentration from this equation is then used in Eq. (7), making sure that $c_{LE} \leq c_{max}$. If $c_{LE} > c_{max}$, then $c_E < c_L$ and the duration is zero.

4. Results

4.1. Estimated probabilities of failure

Table 3 summarizes the probabilities derived from the Monte Carlo simulations and the resulting probabilities of failure for scenarios A and B.

4.2. Failure type I: exceeding critical concentration just downstream the runout deposit

If a landslide occurs, the probability that the Pb concentration in the near field (adjacent to the landslide runout deposit) will reach the critical concentration for aquatic toxicity for fresh water and stay above the critical concentration at least during two weeks and up to two years is 93%. The probability of failure at any given time is 0.28%.

If a landslide occurs, the probability that the Hg concentration in the near field will stay above the critical concentration for aquatic toxicity for fresh water during two weeks is 72%, and 65% that it will stay above the critical concentration during two years, i.e., somewhat lower. The probability of failure at any given time to reach the critical concentrations for Hg for a duration of two weeks is 0.22%, and decreases to 0.20% for a duration of two years.

Thus, given a landslide, and if the contaminated landslide runout deposit remains untreated, there is a very high probability that the Pb and Hg concentration, respectively, will stay above aquatic toxicity levels for fish and other living organisms in fresh water up to at least two years after the event, if no dredging takes place.

4.3. Failure type II: exceeding critical concentration in the estuary

If a landslide occurs, the probability that the Pb concentration in the far field (the estuary) will reach aquatic toxicity levels for fish and other living organisms in marine water and stay at least at that concentration during two weeks is 0.10%. The probability is 0.01% for the concentration to stay at or above the critical value during two years. For Hg, the corresponding probabilities are 0.65% and 0.2%, respectively.

At any given time, the probability that the Pb concentration will reach the critical level for a duration of two weeks is 0.0003% and 0.00003% for a duration of two years. For Hg, the corresponding probabilities are 0.002% and 0.001%, respectively.

Sedimentation of contaminated suspended matter in the river successively decreases the concentration downstream. If a landslide occurs and the contaminated landslide runout deposit remains untreated, there is a small probability that the concentration of Pb and Hg will stay above the aquatic toxicity levels for fish and other living organisms in the marine water in the estuary situated 18 km downstream of the potential landslide. The shorter the distance to the estuary, however, the higher the probability that critical concentration levels will be reached.

4.4. Failure type III: contaminant load to the estuary

The release of contaminants from the landslide runout deposit was also compared to the annual background load at the river mouth. In case of a landslide, the probability is 4.8% to reach 1% of the annual load to the estuary for Pb, and 8.2% for Hg. After 0.5 years, the probability to reach a contribution corresponding to 1% of the annual background load increases to 21% for Pb, and to 26% for Hg. After 2 years, the probability increases to 25.9% for Pb and 31% for Hg. Now, if the contribution from the contaminated landslide runout deposit is instead compared to 50% of the annual background load, the probability is below 5% for both Pb and Hg, compared with 10% of the annual background load to the estuary which gives probabilities up to and above 10%, if the landslide deposit is left in the river.

At any given time, the probability that the runout deposit releases Pb and Hg that corresponds to 1% of the annual background load two weeks after the event is 0.01% (Pb) and 0.02% (Hg). Two years after the event, the probability that the runout deposit releases contaminants corresponding to 1% of the background load is 0.08% for Pb and 0.09% for Hg.

4.5. Sensitivity analysis

In a case study application, the sensitivity of each of the uncertain input variables (i.e., assumptions) is of particular interest. Table 4 shows the rank correlation coefficients as calculated by Crystal Ball for the simulation of different output variables (forecast). The rank correlation coefficient is a measure that includes both how much the uncertainty of an input variable (assumption) affects the uncertainty of the output variable (forecast) and the relationship between input and output in the model.

Looking at the forecast associated with failure for scenario I, i.e., the duration for the concentration to be above the guideline value near the runout deposit (t_L; $x = 0$), the mean concentration of Pb (or Hg) is the (uncertain) variable with the most influence on the output (0.67). The second most influential variables are the erodibility coefficient (−0.23) and the mean river flow (−0.21). This indicates that a higher mean concentration of lead in the runout deposit will give a longer duration with concentrations above the critical level, whereas higher erodibility coefficients and a higher mean river flow will give a shorter

Table 3
Summary of the main results. The table shows the conditional probabilities of exceeding defined criteria derived from the Monte Carlo simulations and the resulting probabilities of failure (P_f). The vertical bar denotes conditional probabilities. For example, the expression "$P_{fI,\,A}\,|\,L$" means the probability of failure of type I, Scenario A, given that a landslide has occurred (or conditioned on that a landslide has occurred), P = Probability, f = failure, L = Landslide, c_0 = concentration of the contaminant of interest, t = time, E = estuary, W = load. Bold figures denote the probabilities of failure given a landslide. The resulting probabilities are given with a maximum of two significant figures because of small variations in results from different Monte-Carlo runs.

Type of failure	Scenario	Time	Mathematical expression		Probabilities	
					Pb	Hg
I. Exceeding critical concentration just downstream the runout deposit during time period X.	**A**	2 weeks	$P_{f.IA}\,\vert\,L$	$P[c_0 > c_{Aq.\ fresh\ w.\ t=1}]$	**94%**	**72%**
			$P_{f.IA}$	$P[c_0 > c_{Aq.\ fresh\ w.\ t=1}] \times P_L$	0.28%	0.22%
	B	0.5 yrs	$P_{f.IB.\ t=2}\,\vert\,L$	$P[c_0 > c_{Aq.\ fresh\ w.\ t=2}]$	**94%**	**71%**
			$P_{f.IB.\ t=2}$	$P[c_0 > c_{Aq.\ fresh\ w.\ t=2}] \times P_L$	0.28%	0.21%
		1 year	$P_{f.IB.\ t=3}\,\vert\,L$	$P[c_0 > c_{Aq.\ fresh\ w.\ t=3}]$	**93%**	**70%**
			$P_{f.IB.\ t=3}$	$P[c_0 > c_{Aq.\ fresh\ w.\ t=3}] \times P_L$	0.28%	0.21%
		2 yrs	$P_{f.IB.\ t=4}\,\vert\,L$	$P[c_0 > c_{Aq.\ fresh\ w.\ t=4}]$	**93%**	**65%**
			$P_{f.IB.\ t=4}$	$P[c_0 > c_{Aq.\ fresh\ w.\ t=4}] \times P_L$	0.28%	0.20%
II. Exceeding critical concentration n in the estuary during time period X.	**A**	2 weeks	$P_{f.IIA}\,\vert\,L$	$P[c_0 > c_{Aq.\ marine\ w.\ t=1}]$	**0.10%**	**0.65%**
			$P_{f.IIA}$	$P[c_0 > c_{Aq.\ marine\ w.\ t=1}] \times P_L$	0.0003%	0.002%
	B	0.5 yrs	$P_{f.IIB.\ t=2}\,\vert\,L$	$P[c_0 > c_{Aq.\ marine\ w.\ t=2}]$	**0.06%**	**0.44%**
			$P_{f.IIB.\ t=2}$	$P[c_0 > c_{Aq.\ marine\ w.\ t=2}] \times P_L$	0.0002%	0.001%
		1 yr	$P_{f.IIB.\ t=3}\,\vert\,L$	$P[c_0 > c_{Aq.\ marine\ w.\ t=3}]$	**0.03%**	**0.33%**
			$P_{f.IIB.\ t=3}$	$P[c_0 > c_{Aq.\ marine\ w.\ t=3}] \times P_L$	0.0001%	0.001%
		2 yrs	$P_{f.IIB.\ t=4}\,\vert\,L$	$P[c_0 > c_{Aq.\ marine\ w.\ t=4}]$	**0.01%**	**0.20%**
			$P_{f.IIB.\ t=4}$	$P[c_0 > c_{Aq.\ marine\ w.\ t=4}] \times P_L$	0.00003%	0.001%
III. Exceeding 1%, 10% or 50% of the total annual load	**A 1%W$_{bg}$**	2 weeks	$P_{f.IIIA.\ 1\%}\,\vert\,L$	$P[W_{t=1} > 1\%W_{bg}]$	**4.8%**	**8.2%**
			$P_{f.IIIA.\ 1\%}$	$P[W_{t=1} > 1\%W_{bg}] \times P_L$	0.01%	0.02%
	A 10% W$_{bg}$	2 weeks	$P_{f.IIIA.\ 10\%}\,\vert\,L$	$P[W_{t=1} > 10\%W_{bg}]$	**0.39%**	**0.93%**
			$P_{f.IIIA.\ 10\%}$	$P[W_{t=1} > 10\%W_{bg}] \times P_L$	0.001%	0.003%
	A 50% W$_{bg}$	2 weeks	$P_{f.IIIA.\ 50\%}\,\vert\,L$	$P[W_{t=1} > 50\%W_{bg}]$	**0.02%**	**0.07%**
			$P_{f.IIIA.\ 50\%}$	$P[W_{t=1} > 50\%W_{bg}] \times P_L$	0.00%	0.000%
	B 1%W$_{bg}$	0.5 yrs	$P_{f.IIIB.\ 1\%.\ t=2}\,\vert\,L$	$P[W_{t=2} > 1\%W_{bg}]$	**21%**	**26%**
			$P_{f.IIIB.\ 1\%.\ t=2}$	$P[W_{t=2} > 1\%W_{bg}] \times P_L$	0.06%	0.08%
	B 10% W$_{bg}$	0.5 yrs	$P_{f.IIIB.\ 10\%.\ t=2}\,\vert\,L$	$P[W_{t=2} > 10\%W_{bg}]$	**5.7%**	**9.3%**
			$P_{f.IIIB.\ 10\%.\ t=2}$	$P[W_{t=2} > 10\%W_{bg}] \times P_L$	0.02%	0.03%
	B 50% W$_{bg}$	0.5 yrs	$P_{f.IIIB.\ 50\%.\ t=2}\,\vert\,L$	$P[W_{t=2} > 50\%W_{bg}]$	**1.1%**	**2.4%**
			$P_{f.IIIB.\ 50\%.\ t=2}$	$P[W_{t=2} > 50\%W_{bg}] \times P_L$	0.003%	0.01%
	B 1%W$_{bg}$	1 yr	$P_{f.IIIB.\ 1\%.\ t=3}\,\vert\,L$	$P[W_{t=3} > 1\%W_{bg}]$	**27%**	**32%**
			$P_{f.IIIB.\ 1\%.\ t=3}$	$P[W_{t=3} > 1\%W_{bg}] \times P_L$	0.08%	0.09%
	B 10% W$_{bg}$	1 yr	$P_{f.IIIB.\ 10\%.\ t=3}\,\vert\,L$	$P[W_{t=3} > 10\%W_{bg}]$	**9.1%**	**14%**
			$P_{f.IIIB.\ 10\%.\ t=3}$	$P[W_{t=3} > 10\%W_{bg}] \times P_L$	0.03%	0.04%
	B 50% W$_{bg}$	1 yr	$P_{f.IIIB.\ 50\%.\ t=3}\,\vert\,L$	$P[W_{t=3} > 50\%W_{bg}]$	**2.4%**	**4.5%**
			$P_{f.IIIB.\ 50\%.\ t=3}$	$P[W_{t=3} > 50\%W_{bg}] \times P_L$	0.01%	0.01%
	B 1%W$_{bg}$	2 yrs	$P_{f.IIIB.\ 1\%.\ t=4}\,\vert\,L$	$P[W_{t=4} > 1\%W_{bg}]$	**26%**	**31%**
			$P_{f.IIIB.\ 1\%.\ t=4}$	$P[W_{t=4} > 1\%W_{bg}] \times P_L$	0.08%	0.09%
	B 10% W$_{bg}$	2 yrs	$P_{f.IIIB.\ 10\%.\ t=4}\,\vert\,L$	$P[W_{t=4} > 10\%W_{bg}]$	**8.6%**	**13%**
			$P_{f.IIIB.\ 10\%.\ t=4}$	$P[W_{t=4} > 10\%W_{bg}] \times P_L$	0.03%	0.04%
	B 50% W$_{bg}$	2 yrs	$P_{f.IIIB.\ 50\%.\ t=4}\,\vert\,L$	$P[W_{t=4} > 50\%W_{bg}]$	**2.1%**	**4.1%**
			$P_{f.IIIB.\ 50\%.\ t=4}$	$P[W_{t=4} > 50\%W_{bg}] \times P_L$	0.01%	0.01%

duration, but with somewhat less impact (i.e., the rank correlation coefficients are lower).

For the forecast associated with failure for scenario II, i.e., the duration for the concentration to be above the guideline value in the estuary ($t_{L,\,E}$; $x = 18{,}000$), none of the uncertain input variables have a large impact. This is because only a few of the runs in the simulations for the given setup result in a time duration above 0 s. In order for the uncertain input variables to have an impact, the setup needs to be different, e.g., the concentration in the runout deposit needs to be much higher, or the guideline value for the estuary needs to be much lower, combined with a low erodibility coefficient and a low mean river flow.

For the forecast associated with failure for scenario III, i.e., the mass deposited in the estuary (M, ΔV_E) after $t = 1$ (i.e., 2 weeks), it appears to be most influenced by the settling velocity (−0.90), followed by the mean river flow (0.31) and the erodibility coefficient (−0.27). Thus, a higher settling velocity and a higher erodibility coefficient will produce less mass deposited in the estuary, whereas a higher mean river flow (i.e., higher shear stress) will produce a larger mass deposited in the estuary. The same is valid for Scenario B (i.e., $t = 2$, 3 and 4).

For the general methodology, it is also of interest to look at the full parameter sensitivity, i.e., to consider the parameters which are not treated as uncertain variables in the case study application. Burgman (2005) suggests the following formula for calculating a sensitivity index, S:

$$S = \frac{\Delta y / y_1}{\Delta x / x_1} = \frac{(y_2 - y_1)/y_1}{(x_2 - x_1)/x_1} \tag{13}$$

where y is the output variable of interest and x the input variable one wishes to investigate the impact of. A value of S close to 0 means that

Table 4
Rank correlation coefficients generated in Crystal Ball. A positive coefficient indicates a direct relationship between the input variable (assumption) and the output variable (forecast), whereas a negative sensitivity indicates an inverse relationship. The highest sensitivity is associated with those rank correlation coefficients that are close to ±1. The highest coefficients are marked with bold numbers, and coefficients lower than ±0.1 are marked with gray numbers.

				Assumption					
				Density of runout deposit	Drag coefficient	Erodibility coefficient	Mean concentration (Pb)	Mean river flow	Settling velocity
Forecast	Time (t)	Distance (x)	Type of failure:	δ_s	C_D	K_τ	$C_{soil,Pb}$	Q	w
t_L	–	0	I, A & B	0.08	-0.03	−0.23	**0.67**	−0.21	**−0.10**
$t_{L,E}$	–	18 000	II, A & B	0.00	0.01	0.03	0.05	0.03	−0.05
$M, \Delta V_E$	1	18 000	III, A	−0.01	0.04	−0.27	0.12	0.31	**−0.90**
$M, \Delta V_E$	2	18 000	III, B	0.00	0.04	−0.28	0.12	0.31	**−0.91**
$M, \Delta V_E$	3	18 000	III, B	0.00	0.04	−0.29	0.12	0.30	**−0.91**
$M, \Delta V_E$	4	18 000	III, B	0.00	0.04	−0.30	0.12	0.30	**−0.91**

there is no sensitivity, $S = 1$ means that the relationship is proportional, and $S > 1$ means that the output is sensitive to that input variable. Positive and negative signs show whether the relationship is direct or inverse.

All input variables in the model are varied individually with all other variables kept constant. Here, the variables are varied +50% and −50% from their original value, i.e., the most likely or mean value, or the fixed value depending on the type of variable. Table 5 shows the results of calculating the sensitivity index (S) for the output variables characterizing the three types of failure (I, II, and III) for time $t = 1$, i.e., 2 weeks: t_L, $t_{L,E}$ and M, ΔV_E.

When all input variables are considered in the sensitivity analysis, the most important parameters are the mean river flow, the water depth, and the deposit width with regard to t_L, i.e., the duration of a concentration to be above a critical value at $x = 0$. All these parameters are related to the river water velocity, which will have an influence on whether the duration is longer or shorter. Thus, a landslide in a slow flowing river will cause the concentrations to stay high for a long time.

Table 5
Calculated sensitivity index S for the output variables (y) for all input variables (x). All calculations for $t = 2$ weeks. The input variables (x) are varied +50% (+) and −50% (−) from their original value. Shaded boxes refer to combinations of variables for which the equations are not valid (when $h_l < h_0 \times 0.5$). The notation n.r. refers to where the output variable is equal to 0 for the given combinations of input variables. Indices above 1 are marked with bold numbers.

		Output variables, y					
		Duration at runout deposit t_L		Duration in estuary $t_{L,E}$		Mass deposition in estuary $M, \Delta V_E$	
Input variables, x	**Symbol**	+	−	+	−	+	−
Settling velocity	w	0.00	0.00	n.r.	n.r.	**−2.00**	**−2837**
Erodibility coefficient	K_T	−0.45	−0.95	n.r.	n.r.	0.99	0.99
Drag coefficient	C_D	−0.45	−0.95	n.r.	n.r.	0.99	0.99
Density, runout deposit	r_s	1.00	1.00	n.r.	n.r.	0.01	0.02
Mean river flow	Q	−0.97	**−3.90**	n.r.	n.r.	**559**	**2.00**
Mean concentration (Pb)	$C_{soil,\,Pb}$	0.32	0.52	n.r.	n.r.	1.00	1.00
Distance from the deposit to the estuary	X_R	0.00	0.00	n.r.	n.r.	**−2.00**	**−2837**
Water depth, mean	h_0	**2.08**	1.59	n.r.	n.r.	**−1.57**	−49.8
Deposit thickness	t_D	0.12	0.34	n.r.	n.r.	3.77	**1.36**
Deposit width (cross the river)	B	**1.80**	**1.36**	n.r.	n.r.	**−2.00**	**−5512**
Deposit length (along the river)	L	0.32	0.52	n.r.	n.r.	1.00	1.00
Pb guideline value, near field	$c_{gv,\,Pb,\,near\,field}$	−0.31	−0.55	n.r.	n.r.	0.00	0.00
Pb guideline value, far field	$c_{gv,\,Pb,\,far\,field}$	0.00	0.00	n.r.	n.r.	0.00	0.00

Again, for the duration of the concentration to be above the guideline value in the estuary ($t_{L,E}$), none of the input variables have a large impact. None of the combinations of input variables can produce a concentration at the estuary above the guideline value, and the time is thus 0 for all combinations – the sensitivity is also zero for all variables. However, if the landslide area is very close to the estuary, it is possible that such a situation could produce concentrations in the estuary above some critical level.

The mass deposited in the estuary after 2 weeks (M, ΔV_E) is influenced most by the settling velocity, the mean river flow, the distance to the estuary, the water depth, and the deposit length. These parameters all have to do with the amount of material that can settle before the estuary. So, a landslide with material having a low settling velocity (i.e., fine material) combined with a high flow velocity (high shear stress), will increase the possibility for a large deposit of contaminated mass in the estuary.

5. Discussion

5.1. Case study

The probability that the contaminated area will slide into the Göta Älv River was calculated to be 0.3% for a 50-year time period, taking into account changes in slope geometry due to river bank and bed erosion. A probability of 0.3% (3×10^{-3}) may seem quite low, but it markedly exceeds the safety target of 10^{-6} for hydrologic events from the American Nuclear Society (1981), as well as the value of 10^{-4} in the risk guideline chart for dam failure by the U.S. Department of the Interior (2011). The Swedish Environmental Protection Agency (Naturvårdsverket) has determined an acceptable risk level in relation to contaminated soil that no >1 person in a population of 100,000 (i.e., 10^{-5}) should suffer from cancer during their lifetime because of exposure to contaminants (Naturvårdsverket, 2009).

A landslide probability of 0.3% means that the probability for the defined failures can at most be equal to (if the probability of all other events is equal to 1) or lower than 0.3%. However, in case of a landslide, it is highly likely for Pb (>90%) and Hg (>60%) that a contaminated landslide runout deposit will cause water concentrations adjacent to the landslide deposit to stay above the water quality guidelines for at least two years, if the landslide deposit is not removed.

The probability that the contaminant contribution from the landslide runout deposit will significantly contribute to the yearly load cannot be neglected. However, depending on the choice of EQS, the probability to exceed a portion of the annual background load varies. The probability that a landslide event at the case study site will contribute to an additional contaminant load to the estuary that corresponds to 1% of the annual background load of Hg is 8% (5% for Pb), even if the landslide deposit is removed after 2 weeks.

The calculations indicate that it will take several years for the contaminated landslide runout deposit to be eroded. Erosion of the deposit is greater in the beginning when the water flow generates high velocities over the deposit, leading to high shear stresses. The more the deposit is eroded the more the velocity decreases and the erosion process slows down. The runout deposit will therefore act as a continuous load until all the contaminated deposit is eroded, and the contaminants have been transported from the deposit to the river and the river mouth. In practice, it means that the contaminants move from a concentrated environment to a diluted environment. The calculations indicate a probability of 47% that it will take >4 years to erode 0.5 m of the runout deposit, and a probability of 50% that it will take >11 years to erode 1 m of the deposit, and even longer for almost all of the deposit to be eroded (approximately 66 years). Although the present description is a simplification of the governing processes, the results suggest that contaminants will be long-lived and available for transportation and exposure to organisms for a long time, if no dredging or other remedial measures are done.

The sensitivity analysis does not give very surprising results; what ends up in the estuary depends mainly on the amount of material that can settle in the river before reaching the estuary, and the duration of any critical concentrations depends on how much contaminants there is in the runout deposit. To reduce the uncertainty in the estimation of the risk, better data on the lead and mercury concentrations in the soil, as well as better data on the settling velocity and the erodibility coefficient, have the highest potential to reduce the uncertainty.

It should be mentioned that the case study site has now undergone a cleanup. The site was one of the most prioritized due to the environmental risk and the landslide risk, combined with the risk for reduced reliability regarding the water supply to the municipal residents.

5.2. Method for estimating long-term ecological risks

The suggested method presented herein is a probabilistic method to estimate the long-term risks to aquatic organisms from contaminated landslide runouts in rivers. For a full risk assessment, a proper description of the consequences to organisms from particle-bound contaminants should be included. With regard to the effects on organisms, it may well be that the particles themselves, regardless of contamination content, also act as stressors and the combination of particles and contamination should thus be considered in the analysis. It is a limitation that the present method only estimates the potential for a risk, and the method should thus be considered a proxy-method. Nevertheless, the method does provide insights into potential long-term ecological risks from landslides bringing contaminated soil or waste into surface waters.

The equations employed describing the landslide deposit, as well as the processes controlling the subsequent erosion of the deposit and the transport downstream of the eroded material, involve significant schematizations. However, the basic physical mechanisms for mobilization and transport of material is included implying that the model should capture the overall behavior of the transport of material from the deposit after the landslide. Since the focus here was on the long-term behavior with a slow release of material from the deposit, the details of the spatial and temporal variations in the river stretch downstream of the deposit were not modeled in detail, but it was assumed that this stretch responded uniformly with respect to the material release at the assumed time scale. By using the advection-dispersion equation the detailed spatial and temporal evolution in the downstream river stretch can be modeled; however, then a numerical approach must be taken that complicates the implementation of the present method, requiring substantially more computational effort. Similarly, the geometry and other properties of the deposit can be made more general or detailed, as well as the modeling of the material release from the deposit, but again a numerical approach is required. This is thus an alteration from the previous study where an ADE was considered relevant for the instantaneous and short-term release, and where an analytical solution could be found.

Overall, the sensitivity analysis including all input variables shows a somewhat different result than the sensitivity data generated by Crystal Ball (Section 4.5). However, for a given area at risk for landslides, several of the input variables would be known, and thus for any specific case, one should investigate which of the uncertain variables that are most influential. The sensitivity analysis of the uncertain variables for the case study shows that the settling velocity becomes an important variable to consider, followed by the contaminant concentration in the runout deposit, the mean river flow, and the erodibility coefficient. The erodibility coefficient is one of the most difficult variables to determine; it is not only site specific but can vary quite a lot within a small area and within small depths. It is difficult to measure the erodibility coefficient, both in the field and the laboratory, hence the parameter will probably stay uncertain, although the range of uncertainty can be reduced. Tests can be performed to decrease the uncertainty regarding the settling velocity as well as the mean concentration in the deposit, which are two critical variables, especially for the calculation of the concentration just downstream the landslide runout deposit. Correlation between parameters is only partly treated in our study, and the assumed correlation is uncertain and assumed to be related to the specific material considered in the landslide deposit.

In this study, we have only investigated potential impact on the water quality from contaminated landslide runout deposits. We have not considered other environmental effects such as flooding of contaminated land from landslide-generated waves, or the erosion of contaminated sediments and bank material that such waves could generate, as well as the damming effects and associated possible dam bursts. These scenarios may very well be necessary to take into account for large landslides.

6. Conclusions

The overall objective of this study was to develop a method to estimate the long-term risks for the aquatic life from the deposition of a contaminated landslide runout in a river. Despite some simplifications, the method provides indications on the potential long-term ecological risks when a landslide brings contaminants into a river. The method can be used as an addition to contaminated site risk assessment, landslide risk analysis, or be a part of an integrated water management plan that considers several possible risks (cascading risks) for the water body in a long-term perspective. If the method presented here is combined with the previous developed method for the instantaneous release of contaminants, the short-term perspective can be considered as well.

The case study indicates that a contaminated landslide runout deposit will release contaminants that most likely will reach fresh water toxicity levels in the near field and for a long period of time (several years), if dredging of the contaminated runout deposit is not carried out. It may theoretically take >60 years for the whole landslide runout deposit to erode without dredging. Long-term Pb and Hg concentrations in the far field estuary are not likely to reach marine water toxicity levels. The probabilities of exceeding a 1%, 10% or 50% contribution to the annual Pb and/or Hg load that enters the estuary are assumed to be moderate to low, depending on how much additional contribution to the annual load that is considered to be a problem in the estuary.

Acknowledgments

The authors would like to thank the Åforsk Foundation (grant number 15-388) in Sweden for the funding of the study, and the Swedish Geotechnical Institute (grant number 1.1-1705-0318) for co-funding. We also would like to thank Ale Municipality for the sharing of data and knowledge. Finally, we thank the reviewers for valuable comments that greatly improved the paper.

Appendix A. Model of contaminant transport from a runout deposit in a river

The runout deposit is assumed to have a rectangular shape with length (L) along the river, spanning the entire width (B) (see Fig. 1). Since the water depth over the deposit (h) is smaller than the depth along the unaffected river (h_o), the velocity will be larger over the deposit, implying a greater capacity for sediment transport. Applying the continuity equation for water flow from a point upstream to the deposit yields (same volume flux per unit width along the river),

$$U = U_o \frac{h_o}{h} \tag{A1}$$

where U_o is the mean velocity in river upstream the deposit where the water depth is h_o and U is the mean velocity over the deposit. The cross section of the river is taken to be rectangular with constant properties. In a more detailed description of the flow over the deposit, the energy equation (or momentum equation) is needed together with the continuity equation, since both h and U are unknown. Applying only the continuity equation assumes that the water surface is horizontal, which is not correct since a dip in the water depth over deposit occurs (if the flow is subcritical upstream the deposit). However, for the model developed here, using Eq. (A1) to obtain U is satisfactory, although it is expected that the model will yield progressively poorer predictions as the water depths over the deposit becomes smaller. Also, the spatial variation in the flow conditions along the deposit is not taken into account, but only a single representative velocity (U) is employed.

Initially, two different sediment transport formulas were employed, one applicable for cohesive sediment and one for non-cohesive sediment, primarily bed load. In both cases the excess shear stress at the bed, that is, the bed shear stress (τ_B) over a critical shear stress for sediment transport (τ_{cr}), determines the transport rate, but the relationships are different. The bed shear stress is calculated from,

$$\tau_B = C_D \rho U^2 \tag{A2}$$

where C_D is a drag coefficient determined by the flow conditions and roughness at the bed and ρ is the water density. In order to describe the erosion of the runout deposit, a continuity equation is employed where the volume change of the deposit is related to the sediment transport from the deposit yielding,

$$L\frac{dh}{dt} = q \tag{A3}$$

where q is the net sediment volume transport per unit width from the deposit. The relationship between q and the mass transport per unit time ($\dot{m}$) is given by,

$$\dot{m} = \rho_s B q \tag{A4}$$

where ρ_s is the sediment density.

In the present study only the transport equation for cohesive material was used (Sanford and Maa, 2001),

$$E = K_T(\tau_B - \tau_{cr}) \tag{A5}$$

where K_T is an empirical transport coefficient and E is the mass transport per unit surface area and time. In the application of Eq. (A5) it is assumed that the material eroded from the bed is suspended in the water column and transported away by the flowing water, not subject to any sedimentation along the runout deposit. Thus, E multiplied by the bed area of the deposit exposed to the flowing water ($A_B = BL$) yields $\dot{m}$, which constitutes the input of material for transport and spreading in the downstream part of the river. Developing Eq. (A5), the volume transport may be expressed as:

$$q = \frac{K_T L}{\rho_s}(\tau_B - \tau_{cr}) \tag{A6}$$

Furthermore, introducing Eq. (A2) for the shear stresses in Eq. (A6) yields,

$$q = C_D K_T L \frac{\rho}{\rho_s}\left(U^2 - U_{cr}^2\right) \tag{A7}$$

Employing this expression for q in the continuity equation for the deposit (Eq. (A3)) gives an evolution equation in terms of the depth over the deposit,

$$\frac{dh}{dt} = C_D K_T \frac{\rho}{\rho_s}\left(U^2 - U_{cr}^2\right) \tag{A8}$$

where U_{cr} is the velocity required for incipient sediment transport obtained from $U_{cr} = \sqrt{\tau_{cr}/\rho C_D}$.

Assuming that the river in its original state is more or less in equilibrium with the flow, the sediment transport would be small or negligible and U_o would correspond to the critical velocity, that is, $U_{cr} = U_o$. Based on this condition, Eq. (A8) can be developed to:

$$\frac{dh}{dt} = C_D K_T \frac{\rho}{\rho_s} U_o^2 \left(\frac{U^2}{U_o^2} - 1\right) \tag{A9}$$

Introducing the continuity equation for water flow (Eqs. (1), (9)) may be written:

$$\frac{dh}{dt} = C_D K_T \frac{\rho}{\rho_s} U_o^2 \left(\frac{h_o^2}{h^2} - 1\right) \tag{A10}$$

This differential equation can be solved analytically with the initial conditions that $h = h_I$, when $t = 0$. The equation may be solved through variable separation and the solution is obtained in implicit form with t as a function of h according to,

$$h_o \ \text{arctanh}\left(\frac{h_o(h - h_I)}{h_o^2 - h h_I}\right) - h + h_I = \lambda U_o t \tag{A11}$$

where a rate coefficient (λ) appears defined by:

$$\lambda = K_T C_D \frac{\rho}{\rho_s} U_o \tag{A12}$$

In the initial phase of deposit erosion, Eq. (A11) may be simplified by using a Taylor series expansion of the arctanh-term yielding:

$$h = \frac{1}{2}(h_I - \lambda U_o t) + \sqrt{\frac{(h_I - \lambda U_o t)^2}{4} + \frac{\lambda U_o t h_o^2}{h_I}} \tag{A13}$$

As long as the term $\lambda t U_o / h_o$ is small, Eq. (A13) constitutes a good approximation to the exact solution given by Eq. (11).

References

Ale Municipality, 2014. Remediation of Surte 2:38, Final Report 2014-03-15 (in Swedish: Sanering av området Surte 2:38).

Alén, C., 1998. On probability in geotechnics. Random Calculation Models Exemplified on Slope Stability Analysis and Ground-Superstructure Interaction. Department of Geotechnical Engineering. Chalmers University of Technology (ISBN: 91-7197-637-x).

Aven, T., 2003. Foundations of Risk Analysis: A Knowledge and Decision-Oriented Perspective. John Wiley & Sons, Ltd.

Aven, T., Renn, O., 2009. On risk defined as an event where the outcome is uncertain. J. Risk Res. 12 (1), 1–11.

Berggren, B., Alén, C., Bengtsson, P.E., Falemo, S., 2011. Method description Probability for landslide: Quantitative model for calculation (*in Swedish*: Metodbeskrivning sannolikhet för skred: Kvantitativ beräkningsmodell). Swedish Geotechnical Institute.

Birsen, N., Kadyrzhanov, K., 2003. Environmental protection against radioactive pollution. Proceeding of the NATO Advanced Research Workshop on Environmental Protection Against Radioactive Pollution. Almati, Kazakhstan, Sept. 2002 (ISBN 1-4020-1789-8).

Burgman, M., 2005. Risks and decisions for conservation and environmental management. Ecology, Biodiversity and Conservation. Cambridge University Press.

Business Vancover, 2015. Landslide releasing toxic metals could indicate site C reservoir issues. (Available at). http://www.policynote.ca/toxic-landslides-into-the-peace-river-continue-add-to-fears-about-impacts-of-site-c-and-fracking/, Accessed date: October 2016.

Carroll, R., Warwick, J., 2016. Modeling the highly dynamic loading of mercury species in the Carson River and Lahontan Reservoir System, Nevada. J. Am. Water Resour. Assoc. 52 (5):1207–1222. https://doi.org/10.1111/1752-1688.12448.

CCME, 2016. Canadian council of ministers of the environment. Canadian environmental quality guidelines summary table. http://st-ts.ccme.ca/en/index.html, Accessed date: December 2016.

Coulthard, T.J., Mackling, M.G., 2003. Modeling long-term contamination in river systems from historical metal mining. Geology 31 (5), 451–454.

Covello, V.T., Merkhofer, M.W., 1993. Risk Assessment Methods: Approaches for Assessing Health and Environmental Risks. Plenum Press, New York.

Dai, F.C., Lee, C.F., Ngai, Y.Y., 2002. Landslide risk assessment and management: an overview. Eng. Geol. 64, 65–87.

Foulds, S.A., Brewer, P.A., Mackling, M.G., Haresign, W., Betson, R.E., Rassner, S.M.E., 2014. Flood-related contamination in catchments affected by historical metal mining: an unexpected and emerging hazard of climate change. Sci. Total Environ. 476–477, 165–180.

Göransson, G., Bendz, D., Larson, M., 2009. Combining landslide and contaminant risk: a preliminary assessment. A study of the Göta Älv river Valley. Sweden. J. Soils Sediments 9, 33–45.

Göransson, G., Larson, M., Bendz, D., Åkesson, M., 2012. Mass transport of contaminated soil release into surface water by landslide (Göta River. SW Sweden). Hydrol. Earth Syst. Sci. 16, 1879–1893.

Göransson, G., Norrman, J., Larson, M., Alén, C., Rosén, L., 2014. A methodology for estimating risks associated with landslide of contaminated soil into rivers. Sci. Total Environ. 472, 481–495.

Göransson, G., Hedfors, J., Ndayikengurukiye, G., Odén, K., 2016. Climate change induced river erosion as a trigger for landslide. The 17th Nordic Geotechnical Meeting Reykjavik Iceland, May 25–28, 2016.

Göta älvs vattenvårdsförbund, 2015a. Report on Watercourse Control 2015 (*in Swedish*: Rapport Avseende Vattendragskontroll 2015). Göta Älv River Water Conservation Association (Available at). http://gotaalvvvf.org.

Göta älvs vattenvårdsförbund, 2015b. Facts About Göta Älv River. A Description of Göta Älv and its River Basin Downstream Lake Vänern (*in Swedish*: Fakta om Göta älv. En Beskrivning av Göta älv och dess Avrinningsområde Nedströms Vänern 2015). Göta Älv River Water Conservation Association (Available at). http://gotaalvvvf.org.

Institute of Nuclear Physics NNC RK, 2003. Nuclear science and its application. Proceeding of the Second Eurasian Conference, Volume IIII: Application of Nuclear Technologies in Industry, Medicine and Agriculture, Environmental Protection against Radioactive Pollution. Almaty, Republic of Kazakhstan, 2002 (ISBN 9965-9214-6-6).

IPCC, 2014. Annex II: glossary. In: Mach, K.J., Planton, S., vons Stwchow, C. (Eds.), Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. IPCC, Geneva Switzerland, pp. 117–130 ([Core Writing Team, R.K. Pachauri and L.A. Meyers (eds.)]).

ISO, 2009. Risk management – principles and guidelines. ISO Standard 31000:2009. International Organization for Standardization.

Johnston, S., Rose, A., Burton, E., Webster-Brown, J., 2015. Landslide-induced iron mobilization shapes benthic accumulation of nutrients, trace metals and REE fractionation in an oligotrophic alpine stream. Geochim. Cosmochim. Acta 148:1–22. https://doi.org/10.1016/j.gca.2014.10.022.

McCahon, C.P., Carling, P.A., Pascoe, D., 1987. Chemical and ecological effects of Pennine peat-slide. Environ. Pollut. 45, 275–289.

Meyer-Peter, E., Mueller, R., 1948. Formulas for bed-load transport. Report on the 2nd Meeting International Association Hydraulic Structure Research, pp. 39–64 (Stockholm, Sweden).

Motamedi, M., 2013. Quantitative Landslide Hazard Assessment in Regional Scale Using Statistical Modeling Techniques. (Dissertation). The Graduate Faculty the University of Akron.

Naturvårdsverket, 2009. Guideline Values for Contaminated Soil: Model Description and Guidance. (In Swedish: Riktvärden för förorenad Mark. Modellbeskrivning och Vägledning). Swedish Environmental Protection Agency (Report no 5976).

New Internationalist, 2005. Toxic time bomb. Issue 382. (Available at). https://newint.org/features/2005/09/01/toxic/, Accessed date: October 2016.

NRC, 1996. Understanding risk. Informing Decisions in a Democratic Society. National Academy Press, Washington D.C.

Odén, K., Bergdahl, K., Löfroth, H., Göransson, G., Jönsson, Å., Kiilsgaard, R., Öberg, M., 2017. Mapping landslide risks in a changing climate – Development of a simplified methodology. Chapter 49. (2017). In: Thakur, V., L'Heureux, J.-S., Locat, A. (Eds.), Landslide in Sensitive Clays, From Research to Implementation. Springer International Publishing AG 2017, pp. 571–580.

Otago Daily Times, 2017. US landslide crews face toxic challenge. (Available at). https://www.odt.co.nz/news/world/us-landslide-crews-face-toxic-challenge, Accessed date: October 2016.

Petley, D., 2013. A catastrophic landslide-induced oil spill in Ecuador. AGU Blogosphere. (Posted June 13, 2013, by Dave Petley). http://blogs.agu.org/landslideblog/2013/06/12/a-catastrophic-landslide-induced-oil-spill-in-ecuador-including-two-shocking-videos-of-the-pollution/, Accessed date: September 2017.

Policynote, 2016. Toxic landslides into the Peace River continue, add to fears about impacts of Site C and fracking. (Available at). http://www.policynote.ca/toxic-landslides-into-the-peace-river-continue-add-to-fears-about-impacts-of-site-c-and-fracking/, Accessed date: October 2016.

Rhoades, E., O'Neal, M., Pizzuto, J., 2009. Quantifying bank erosion on the South River from 1937 to 2005, and its importance in assessing Hg contamination. Appl. Geogr. 29 (1): 125–134. https://doi.org/10.1016/j.apgeog.2008.08.005.

Rowan, J.S., Barnes, S.J.A., Hetherington, S.L., Lambers, B., 1995. Geomorphology and pollution: the environmental impacts of lead mining, Leadhills, Scottland. J. Geochem. Explor. 52, 57–65.

SafeLand, 2011. Guidelines for landslide susceptibility, hazard and risk assessment and zoning. Living With Landslide Risk in Europe: Assessment, Effects of Global Change, and Risk Management Strategies (Deliverable D2.4. EU 7th Framework Programme).

Sanford, L.P., Maa, J.P.Y., 2001. A unified erosion formulation for fine sediments. Mar. Geol. 179, 9–23.

Schreier, H., Lavkulish, L., 2015. Cumulative effects of the transport of asbestos-rich serpentine sediments in the trans-boundary Sumas Watershed in Washington State and British Columbia. Can. Water Resour. J. 40 (3):262–271. https://doi.org/10.1050/07011784.2015.1051495.

Seneviratne, S.I., Nicholls, N., Easterling, D., Goodess, C.M., Kanae, S., Kossin, J., Luo, Y., Marengo, J., McInnes, K., Rahimi, M., Reichstein, M., Sorteberg, A., Vera, C., Zhang, X., 2012. Changes in climate extremes and their impacts on the natural physical environment. In: Field, C.B., Barros, V., Stocker, T.F., Qin, D., Dokken, D.J., Ebi, K.L., Mastrandrea, M.D., Mach, K.J., Plattner, G.-K., Allen, S.K., Tignor, M., Midgley, P.M. (Eds.), Managing the Risks of Extreme Events and Disasters to Advance Climate Change Adaptation. A Special Report of Working Groups I and II of the Intergovernmental Panel on Climate Change (IPCC). Cambridge University Press, Cambridge, UK, and New York, NY, USA, pp. 109–230.

SGI, 2012. Landslide risks in the Göta River valley in a changing climate. Final report, Part 2 – Mapping. The Göta River investigation 2009–2011. Swedish Geotechnical Institute.

Ströberg, A., Eberg, K., Jarsjö, J., Frampton, A., 2017. Contaminated area instability along Ångermanälven River, northern Sweden. Enivron. Monit. Assess. 189:118. https://doi.org/10.1007/s10661-017-5839-0.

Swartjes, F., 2011. Introduction to contaminated site management. In: Swartjes (Ed.), Dealing with Contaminated Sites: From Theory towards Practical Application. Springer, pp. 3–89 (Chapter 1).

The American Nuclear Society, 1981. Working Group ANS-2.8. American National Standard for Determining Design Basis Flooding at Power Reactor Sites. American Nuclear Society, La Grange Park. IL (February 1981).

The U.S. Department of the Interior, 2011. Dam Safety Public Protection Guidelines. A Risk Framework to Support Dam Safety Decision-Making (Interim report).

Torgoev, I., Omorov, B., 2014. Mass movement in the waste dump of high-altitude Kumtor goldmine (Kyrgyzstan). Proceedings of World Landslide Forum 3. 2–6 June 2014 Beijing.

UNISDR, 2009. UNISDR Terminology on Disaster Risk Reduction. United Nation International Strategy for Disaster Reduction.

UN-SPIDER, 2017. Environmental effects of possible landslide catastrophe in the areas of radioactive waste warehousing in Kyrgyzstan (central Asia). (by Torgoev I, Aleshin Yu, Ashirov G.). http://www.un-spider.org/sites/default/files/2-Landslide_Impacts_%28Institute%20of%20Physics%20of%20Kyrgyzstan%29.pdf, Accessed date: December 2017.

USGS, 2004. Landslide types and processes. Fact Sheet 2004-3072. U.S. Department of the Interior, U.S. Geological Survey.