



**Geochemical controls on the partitioning and hydrological transport of metals**

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# Geochemical controls on the partitioning and hydrological transport of metals in a non-acidic river system

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## Abstract

The speciation of metals, i.e. in which chemical form they occur, controls their mobility, bioavailability and toxicity. The overall objective of this study is to extend the knowledge on the spreading of metals in non-acidic river systems; this knowledge is currently much more limited than the knowledge on metal behavior under acidic conditions that for instance are found in acid mine drainage systems. We combine novel measurements of metal spreading under distinctly high-pH conditions (up to 9.6) in the Tuul River at the Zaamar Goldfield mining site (Upper Lake Baikal Drainage Basin, Mongolia) with a geochemical modelling approach (Visual MINTEQ). Total mass flows of several metals (Al, Cd, Fe, Mn, Pb and V) showed net increases over the mining site, with metals in suspension generally dominating the total export from the site. Model results showed that a main difference from acid mine drainage geochemistry is that the prevailing high pH causes precipitation of ferrihydrite and gibbsite, which removed between 90 to 100 % of Fe and Al from solution. This notably influenced the behavior of As, Pb and V since their solubilities are controlled by sorption onto ferrihydrite. The combined effects from such geochemical processes (precipitation, sorption) hence explain the high impact of suspended transport to total transport under high pH conditions. Arsenic furthermore showed dissolved concentrations above health risk-based guideline values in several locations and can thus be of main toxic concern in the upper Lake Baikal Drainage Basin. Moreover, present modelling showed that in particular the solubility of Fe, Pb and Zn can increase considerably as DOC concentrations increase due to metal-organic complexation. In high pH systems, seasonality of DOC concentrations can therefore have a major influence on the spreading and toxicity of these metals, as can DOC trends caused by land use change. Present results also suggest that the behavior of Cr, Cu and Mo would be much better understood if a dependable adsorption database for hydroxyapatite could be developed.

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## 1 Introduction

Metals can become exposed to the environment through natural processes, such as weathering of soil and bedrock, and through anthropogenic processes, such as mining, industry and agriculture. They can enter aqueous systems through wash out from surface soils, diffuse groundwater inflow, metal mobilization from enriched sediments, leaching from agricultural areas and mine tailings, catastrophic tailings dam failure and discharge of industrial and mining effluents (e.g. Hudson-Edwards, 2003; Macklin et al., 2006; Mighanetara et al., 2009; Inam et al., 2011). Material mobilization via bank and bed erosion under conditions of geomorphic adjustment can dominate pollutant transport under certain conditions (Lewin and Macklin, 1986; Chalov et al., 2014). Once metals are released into aquatic systems they persist in the environment, since they are non-degradable. Metals held in alluvial stores may therefore continue to pose a threat to ecosystems, long after their initial release, and alluvial sediments may possibly constitute the most significant source of catchment heavy metal pollution (Dennis, 2005; Alexeevsky et al., 2013).

Mounting evidence over the last decades have shown that the speciation of metals (i.e. in which chemical form they occur) controls their mobility, bioavailability and toxicity (e.g. Tessier and Campbell, 1979; Tack and Verloo, 1995; Tipping, 1998; Fytianos, 2001; Landner, 2005). The dissolved form, especially the dissolved inorganic fraction, is considered to be the most toxic form of the majority of metals (e.g. Gundersen and Steinnes, 2001; Nystrand et al., 2012), since it can readily be taken up by organisms and biota (e.g. Törnqvist et al., 2011; Raguz et al., 2013). Many chemical factors and parameters need to be accounted for in determining the speciation, such as pH, redox conditions, oxidation state of the metal and available surfaces for adsorption (e.g. Palleiro et al., 2013). An advantage of equilibrium speciation models is the possibility to predict dominating forms and evaluate controlling mechanisms, such as sorption and precipitation reactions (Lund et al., 2008). Frequently used speciation models include WHAM (Tipping, 1994), PHREEQC (Parkhurst and Appelo, 1999) and Visual

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MINTEQ (Gustafsson, 2009), which are used for both groundwater and surface water applications (e.g. Tipping and Lawlor et al., 1998; Butler et al., 2004; Gustafsson et al., 2009; Korfali and Davies, 2004; Wällstedt et al., 2010; Nystrand et al., 2012). They have the ability to represent surface complexation reactions and binding to organic matter, which are all central to metal transport and partitioning between the water and sediment phase (e.g. Butler et al., 2008).

Although there are many studies on speciation modeling of metals in aqueous systems (e.g. Eary, 1999; Butler et al., 2004; Balistrieri et al., 2007; Gustafsson et al., 2009; Nystrand et al., 2012), the majority of them have focused on metal behavior under acidic or near neutral conditions rather than metal behavior under high pH conditions. Some exceptions are for example a study by Sjöstedt et al. (2009) who measured and modeled the speciation of Al, As and Mo in Swedish lakes that had been limed to near neutral pH and a study by Moldovan and Hendry (2005) who modeled the speciation of As, leaching from uranium mining, up to pH 11. In addition to speciation modeling studies, others have highlighted pollution problems of metals in non-acidic systems (see e.g. Tarras-Wahlberg et al., 2000; Sjöblom et al., 2004; Grosbois et al., 2009; Zak et al., 2009), often showing high metal concentrations in particles but low concentrations in the dissolved form, hence indicating that geochemical controls and dominating processes might differ from well-studied acidic pollution sources.

Furthermore, understanding geochemical controls in combination with physical transport of metals could greatly enhance metal transport and fate predictions (Foster and Charlesworth, 1996; Destouni et al., 2010; Persson et al., 2011). For example, geochemical processes (e.g. adsorption) influence whether or not certain physical processes (e.g. suspended sediment transport; Chalov et al., 2012) will be important for the overall transport of metals. There are however relatively few studies that address these combined effects of physical and chemical controls on metals in aqueous systems (e.g. Caruso et al., 2008; Malmström et al., 2008).

The overall objective of this study is to extend the knowledge on the spreading of metals in non-acidic river systems. We combine novel site specific measurements with

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a geochemical modelling approach (Visual MINTEQ), specifically aiming at (i) evaluating the performance of using this modeling approach by comparing observations and model predictions, (ii) identifying dominating solids and predict the capacity of these solids to sorb and/or co-precipitate other metals that are transported in suspension and (iii) identify dominant controls keeping metals in solution, such as soluble complexation.

As an application example, we consider the distinctly high-pH conditions (Pietron, 2012; Thorslund et al., 2012; Chalov et al., 2012, 2014) of the Tuul River, at Zaamar Goldfield, in Mongolia. It is the largest gold mining site within the Tuul River basin, comprising a widespread source zone. The Tuul River connects to the Orkhon–Selenga River system and is a highly polluted river in the upstream Mongolian part of the transboundary Russian–Mongolian Lake Baikal drainage basin. The many mining companies within Zaamar Goldfield, as well as illegal mining, is considered to have serious impact on the water quality within the river basin (Altansukh and Davaa, 2011; Chalov et al., 2012; Thorslund et al., 2012) and could possibly have an impact on downstream Selenga river and Lake Baikal. Although total (suspended and dissolved) concentrations and mass flows of several metals within this system were high, dissolved concentrations were generally low. In the downstream regions of the Lake Baikal drainage basin however, increasing metal pollution of both sediments and biota has been reported (e.g. Rudneva et al., 2005; Khazheeva et al., 2006). Investigating controls on metal partitioning between the sediment and water phase is thus of particular importance for this system, since metals originating from Zaamar Goldfield could potentially be transported all the way to the vicinity of Lake Baikal that hosts a unique ecosystem and comprises a huge water resource.

## 2 Site description

The mining areas of Zaamar Goldfield are located along the Tuul River valley, between Latitude: 48°17′50″ N and Longitude: 104°24′65″ E (Fig. 1) (AATA, 2008),

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approximately 600 km upstream of the Lake Baikal inlet (AATA, 2008). The Tuul River is a tributary to the Selenga River, a transboundary river between Mongolia and Russia, which is by far the largest river draining into Lake Baikal (Lee et al., 2006), discharging about 30 km<sup>3</sup> of water and 3.5 million tons of sediments annually into the lake area.

5 Before the Selenga River flows into Lake Baikal there is the Selenga delta, covering an area of about 1200 km<sup>2</sup> (USGS, 2011).

Annual average discharge of the Tuul River is 27 m<sup>3</sup> s<sup>-1</sup> (1945–2007), but maximum discharges in the summer period can reach values of around 80 m<sup>3</sup> s<sup>-1</sup>, whereas decreasing discharges are marked throughout the coldest months of the year (November–  
10 March). After the junction of the Tuul River with the Orkhon River, discharges on average increase by an order of magnitude and in the downstream Selenga River, annual average discharge reaches just over 1000 m<sup>3</sup> s<sup>-1</sup> (GEMStat, 2011; MCA, 2011). The average yearly precipitation in the region is between 200 and 250 mm. The climate of the region is semi-arid with warm and dry summers (averaging temperatures of  
15 20 °C), although intense rainfalls do occur, and cold winters (averaging temperatures of –20 °C) (AATA, 2008).

The geology of the area includes sedimentary, igneous, and metamorphic rock formations, such as sedimentary sand and siltstones, igneous gabbros, and metamorphic schists. The region is also naturally consisting of calcium bicarbonates to a large degree (Altansukh and Davaa, 2011) and there are strong interactions between surface  
20 waters and groundwater, due to extensive areas of alluvial unconfined aquifers in the river valleys (Zandaryaa et al., 2008). This impacts the Tuul River quality and quantity, which is fed by groundwater inflows with high water-rock interactions; affecting the pH values throughout Zaamar Goldfield to be high (Lee et al., 2006; Zandaryaa et al.,  
25 2008).

Zaamar Goldfield is the largest gold mining site within the Tuul basin and the extensive source zone contains both alluvial and hard rock mining that stretches for about 45 km along the floodplain of the Tuul River (Lee et al., 2006). The many mining activities over widespread areas and the lack of control of tailing and leakage from settling

ponds contribute to both diffuse and direct pollution of soil, groundwater and surface water (Zandaryaa et al., 2008). High levels of metals have been measured in the Tuul River (Lee et al., 2006; AATA, 2008) and previous mass balance quantifications have shown a net increase in metal loading over the mining zone, especially of metals in suspension (Thorslund et al., 2012).

### 3 Methods of data collection and analysis

#### 3.1 Field methods and sample collection

Water samples were collected from the Tuul River at five locations around Zaamar Goldfield during two field campaigns; one in June 2012 and one in September 2013. Information about the sampling locations, including coordinates, are shown in Table 1 and Fig. 1. Relatively low water discharges were observed during June 2012 campaign ( $Q = 13\text{--}15\text{ m}^3\text{ s}^{-1}$ ), whereas significant flooding with floodplain inundation was documented during September 2013 field campaign ( $Q = 45\text{--}52\text{ m}^3\text{ s}^{-1}$ ). The hydrometeorological conditions also varied between the seasons, e.g. relatively dry weather during the 2013 campaign and abundant rainfalls in the 2012 campaign.

The sampling procedure included grab sampling of water in plastic polypropylene bottles (500 mL), directly from the river, just below the surface. All sampling bottles were rinsed with the selected water before collecting the sample for analysis. The bottled water was then transferred into two high-density polypropylene test tubes (10 mL); one unfiltered sample and one filtered sample (filtered through a  $0.20\text{ }\mu\text{m}$  pore membrane filter). In the 2013 campaign, three replicates for both filtered and unfiltered samples were collected. All test tubes were rinsed thoroughly with the sampled water before collecting the sample for analysis. The samples were acidified (1 %) with concentrated  $\text{HNO}_3$  (65 %) to conserve them for analysis and prevent precipitation of metals.

Temperature and pH were measured in situ, directly in the river, using a Hannah Instrument HI 9828 meter (2012) and HI 98108 (2013). The meters were calibrated with

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pH 7 and 10 solutions before each measurement campaign and then recalibrated during the campaigns. In the 2012 campaign, alkalinity was measured at Moscow State University using the standard titration method, while in the 2013 campaign it was measured using a Total Alkalinity test kit (HI 38014) directly in field.

5 Additionally in the 2013 campaign, an extended sampling procedure including samples for total and dissolved organic carbon (TOC/DOC) were collected into 30 mL high-density polypropylene test tubes (three replicates). The DOC samples were filtered with 0.22  $\mu\text{m}$  pore membrane filters. All samples were then acidified (1 %) with concentrated acid  $\text{HNO}_3$  (65 %) and wrapped in aluminum foil to prevent light from penetrating  
10 it. These procedures were all made directly in the field. Samples for anion ( $\text{F}^-$ ,  $\text{Cl}^-$ ,  $\text{NO}_3^-$ ,  $\text{SO}_4^{2-}$ ,  $\text{Br}^-$ ) analysis were also sampled in 2013 and collected into high-density polypropylene test tubes of 10 mL (three replicates) without further preservation.

### 3.2 Analytical methods

All samples discussed in the present paper were analysed at Stockholm University. Metal concentrations were analysed with inductive coupled plasma optical emission spectrometry (ICP-OES) with the Thermo iCAP 6500 Duo analyser, to determine the contents of; Al, As, B, Ba, Ca, Cd, Co, Cr, Cu, Fe, K, Mg, Mn, Mo, Na, Ni, P, Pb, S, Si, Sn, Sr, Ti, V and Zn. The analysis of samples followed standard procedures using a micro concentric nebulizer and in some cases an ultra sonic nebulizer, CETAC USN  
15 U5000AT+ in order to get better detection limits. Eleven metals were selected from the analytical result dataset for further interpretation and modeling purposes; Al, As, Cd, Cr, Cu, Fe, Mn, Mo, Pb, V and Zn. They were selected because these metals are often found at mining sites (e.g. Hudson-Edwards, 2003) and because of their potential toxicity to humans and biota. Mercury (Hg) is a metal that is often associated with  
20 mining activities but was here excluded since it requires different analytical procedures from those performed here, i.e. fluorescence method. Furthermore, previously reported measurements show repeatedly that concentrations have been low and in many cases below detection limits in recent (2006–2011) years (Thorslund et al., 2012). Total and  
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dissolved organic carbon (TOC/DOC) were analysed using an NPOC (non-purgeable organic carbon) measurement using the analyzer Shimadzu TOC-L CPH. Major anion analysis were performed by liquid chromatography using the IC20 Dionex, following Swedish standard procedures (EN-ISO 10304-1:2009).

### 3.3 Discharge measurements and mass flow estimates

Discharge was measured at the sampling locations along the Tuul River, at the same place and time as water samples were collected for chemical analysis. Measurements were made by wading, by boat or from bridges (depending on the depth of each river section). Velocities were measured with a hydrometric propeller (ISP-1) at 0.6 depth (from the surface) of each width increment. When the depth exceeded 1.5 m an Acoustic Doppler Profiler (ADP) was used, which measures water level and velocity at horizontal profiles across a channel. The discharge of each cross-section was calculated according to Eqs. (1) and (2)

$$Q = \sum_{1}^{n} V_n \cdot A_n \cdot k \quad (1)$$

where  $V_n$  is the velocity of each width increment,  $A_n$  is the area of each rectangular subsection, using the Trapezoidal rule;

$$A_n = \frac{d_n + d_{n-1}}{2} \cdot (b_n - b_{n-1}). \quad (2)$$

The coefficient  $k$  in Eq. (1) is used when the expression is applied to non-steep banks; here  $k$  is equal to 0.7 (Bykov and Vasil'ev, 1972). In Eq. (2)  $d$  is the depth of each subsection and  $b$  is the distance from the bank.

The discharges,  $Q$ , were multiplied with concentrations,  $C$ , according to Eq. (3) to yield estimates of dissolved and total metal mass flows,  $M_f$ , over the five cross-sections along the Tuul River.

$$M_f = C \cdot Q \quad (3)$$

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The simultaneous measurements of pollutant concentrations and discharges were made to get snapshot values of loads along the river reach and over Zaamar Goldfield.

### 3.4 Modeling approach

The geochemical equilibrium model Visual MINTEQ vers. 3.0 (Gustafsson, 2011) was used to model the speciation of selected metals. This model calculates the chemical composition of various inorganic ions in aqueous systems, in a single point (e.g. a water sample), under the assumption of a chemical equilibrium state.

Input variables were the following; total concentrations of all elements analysed (see Table 2 for chosen oxidation states), major anion concentrations, pH, alkalinity ( $\text{HCO}_3^-$ ), temperature,  $p_e$  (redox potential) and DOC (dissolved organic carbon) for each sampling point. Studies (e.g. Nystrand et al., 2012) have shown that modeling results are more accurate if they are based on concentrations of a relatively high number of elements in the input data, since this gives a better representation of the chemistry in the water sample (ionic strength and complex binding is then better represented) and thus decrease uncertainties in model calculations. Concentration data of all elements determined in the ICP-OES analysis were therefore used as input variables (see Sect. 3.2). However, in the subsequent sections, the model outputs for the eleven selected metals are presented and furthered discussed.

To test the impact of pH on the speciation, model runs for each sample were made for two different acidic pH conditions; pH 3.5 and pH 6.

DOC and major anions were not measured during the field campaign in 2012. Thus, for the modeling of the 2012 data, DOC values from the 2013 campaign were used. In addition to this, a 50% increased DOC value was modeled for the 2012 data, to investigate the sensitivity to changing DOC concentrations. Since Sulfate ( $\text{SO}_4^{2-}$ ) and Chloride ( $\text{Cl}^-$ ) concentrations were not measured in the 2012 campaign they had to be approximated. Sulfate concentrations were assumed to equal half of measured Sulfur

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(S) concentrations and Chloride concentrations were set to equal measured Sodium (Na) concentrations.

The redox potential (pe) was estimated according to Eq. (4) for both years, which is assumed representative for well oxygenated waters (Stumm and Morgan, 1996).

$$pe = 20.77 + \frac{1}{4} \log pO_2 - pH \quad (4)$$

where  $pO_2$  was assumed to equal 0.2 atm (i.e. the  $O_2$  content in the atmosphere is 20.95 %), similar to what Wällstedt et al. (2010) used in their study.

The solids that were allowed to precipitate when their respectively solubility constant was exceeded are shown in Table 3. These were chosen based on (i) knowledge about common solids for the present geological conditions at the study site, and (ii) model outputs of “saturation indices”, indicating which minerals could precipitate, based on site specific data and thermo dynamical calculations.

Adsorption to ferrihydrite and gibbsite was estimated using the DLM (diffuse layer model) according to Dzombak and Morel (1990) and Karamalidis and Dzombak (2010) respectively. The DLM is a surface complexation model that simulate the distribution of each metal ion between the dissolved form and adsorbed to a solid (Gustafsson et al., 2009). Both of these minerals were selected due to their high measured particulate concentrations and their sorption capacities. Published data on the solubility of gibbsite varies and is mainly a function of pH, sulfate, fluoride and DOC concentrations. The main difficulty lies in that it is hard in natural systems to separate amorphous  $Al(OH)_3$  from gibbsite. Because of this difficulty the solubility of Al is restricted and its database in Visual MINTEQ is more limited than for ferrihydrite. ferrihydrite is an iron mineral which is a frequently used solid for adsorption in Visual MINTEQ and the adsorption database for this solid is thus extensive. Ferrihydrite is well known for its metal retention capacity due to large surface sites for adsorption and has been shown to correlated well with measured data (e.g. Sjöstedt et al., 2010; Nystrand et al., 2012; Tiberg et al., 2013). Adsorption constants for Vanadium adsorption to ferrihydrite was

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change according to Wällstedt et al. (2010). All other parameter values in this menu were kept on default.

For simulation of complexation of the metals with dissolved organic matter the “Stockholm humic model” (SHM) was used. This model uses a discrete-site approach to represent the metal-humic complexation, similar to the model described by Tipping (1994). The default ratio of 1.65 between active DOM (i.e. the part of the organics that binds to the metals) and DOC (dissolved organic Carbon) was assumed representative and thus not changed. All of the DOM was assumed to be fulvic acid, based on earlier studies (e.g. Khai et al., 2008; Sjöstedt et al., 2010; Wällstedt et al., 2010; Gustafsson et al., 2014), which is also set as default in the model.

## 4 Results

### 4.1 Tuul River water chemistry

Geochemical parameter values, used for the model input, representative for the five sampling locations along the Tuul River reach (T5–T6a) are shown in Table 4. The pH shows in all cases alkaline conditions, with values ranging between 9.6 (T5a) to 8.5 (T6) in the 2012 campaign and between 9.2 (T5a) to 7.6 (T5a) in the 2013 campaign. DOC concentrations vary between 10–12 mg L<sup>-1</sup> throughout the measured river locations. TOC concentrations were on average only a few percent higher than DOC concentrations, hence dissolved organic matter is dominating the total organic matter composition. Estimated alkalinity values are very high, especially from field measurements in 2013. However, changing the alkalinity values in the model input did not change the speciation of selected metals, indicating low sensitivity for this parameter.

Dissolved and total riverine concentrations for the selected metals in the 2012 and 2013 campaigns are shown in Fig. 2a–k. In case of no detectable results, the midpoint of the possible concentration range is shown in a lighter color for illustrative purposes (see Table A1 for DL values). The results show that concentration levels are generally

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in the same order of magnitude during both snapshot measurements (2012 and 2013), with the exception of total concentrations of Al, Cd and Mn which are up to one order of magnitude greater in the 2012 campaign. The differences between particulate and dissolved concentrations are greatest for Al, Fe and Mn with particulate concentrations being on average 178, 35 and 25 times greater than dissolved concentrations in the 2012 campaign respectively 42, 18 and 5 times greater in the 2013 campaign. As, Cd, Cr, Cu and Mo, have dissolved concentrations within the same order of magnitude as their total concentrations. V and Zn show varying behaviour, both having higher dissolved and particulate concentrations depending on sampling point. Overall, total concentrations show a net increase over the site, with increased values at the most downstream point (T6a) compared to the upstream points (T5, T5a), with the exception of Zn and Mo. Several metals (Al, As, Fe and Mn) show total (unfiltered) concentrations that exceeds WHO and Mongolian health risk-based guideline values (Table B1). As additionally shows dissolved (filtered) concentrations exceeding these health risk-based guideline values.

## 4.2 Riverine mass flows

Dissolved and total (dissolved + suspended) mass flows at the five locations along the Tuul River from the campaigns in 2012 and 2013 are shown in Fig. 3a–k (note the different scales on the y-axis). The first location (T5) is not included in the 2013 campaign due to discharge data missing. Taken together, the majority of metals from both campaign years show net increasing total mass flows over Zaamar Goldfield, with values being higher at the most downstream point (T6a) than the most upstream (T5, T5a) points. Due to above mentioned high dissolved and total concentration differences, differences between dissolved and total mass flows are generally high, with total mass flows being up to two orders of magnitude higher than dissolved mass flows, especially for Al, Fe and Mn. Similarly, dissolved mass flows of As, Cd, Cr, Cu and Mo are generally of the same magnitude as total mass flows. Total mass flows of most metals

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Increasing the DOC concentration by 50 % from measured values in 2012 resulted in predicted increased overall dissolution of all investigated metals, with greatest impact on Fe, Pb and Zn solubility; up to 10 %. This increase in DOC concentrations correspondingly lead to predicted increased Fe-organic complexation, with a few percent.

## 5 Discussion

The main chemical parameter controlling solubility of metals in aqueous systems is often considered to be pH (e.g. Bourg and Loch, 1995). Agreement between the present observations of high concentrations of metals in suspension and predicted precipitation and/or sorption controls, show that dissolution is generally limited for several metals under the alkaline conditions that prevail at Zaamar Goldfield. Through the scenario modelling where pH was altered from actual site conditions, it becomes apparent that there is a strong pH dependence of several metals in this system. The scenario modelling at pH 3.5 represents pH conditions common at many well studied acid mine drainage sites (e.g. Elliot et al., 1998; Saria et al., 2006). Under such conditions, practically all metals are dissolved. Model predictions at the pH 6 scenario compared to site conditions also changed the speciation of the metals, however with higher sensitivity for some (Zn, V, Pb and Fe) than for others (Al, As, Cd). The substantial dissolution of Zn between site conditions and pH 6 suggest that it is very sensitive to fluctuations in pH. This agrees with results from for instance Chen et al. (1997), who also showed Zn to be pH sensitive, with a significantly decreasing solubility above pH 6.5, where all dissolved Zn was removed from solution due to adsorption.

Adsorption is a main process responsible for limiting dissolved forms of metals in natural systems (e.g. Zhao et al., 2011). A high affinity of “divalent base metals”, including Cd, Cu, Pb and Zn, to adsorb to solid phases, such as Al and Fe-hydroxides is common (e.g. Early, 1999). In the here modeled system, adsorption to Al (gibbsite) was predicted to limit Cd and Zn solubility, with highest impact on Zn and minor influence on Cd. Further, as Pb was predicted to adsorb to gibbsite instead of ferrihydrite where Al

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concentrations reached double Fe concentrations this solid could also control Pb solubility. Studies on the adsorption to gibbsite under high pH conditions are not abundant, however Weerasooriya et al. (2001) showed sorption of Pb to gibbsite, in the same range as predicted here, with up to 50 % sorption to gibbsite at pH 9, supporting our results. Otherwise, sorption of metals to ferrihydrite is commonly reported (e.g. Shultz et al., 1987; Jung et al., 2005; Sanchez-España et al., 2006) and was here predicted to be the main solid for adsorption of As, Pb and V. The degree of sorption of these metals were all predicted to increase as the input pH was changed to 6, suggesting these metals to be more soluble at the non-acidic site conditions than at pH 6. However, results from the predictions at pH 3.5, showing complete dissolution of previously sorbed metals, indicate that sorption is not an active control under acidic mine drainage conditions. Several other studies have shown highest adsorption of these metals to ferrihydrite at circum-neutral pH. For example, Sanchez-España et al. (2006), showed 90–99 % adsorption of As and Pb to ferrihydrite within the pH range 7–8. Further, Jung et al. (2005) indicated 95 % adsorption of Cd, Pb and Zn onto ferrihydrite at pH 7. This suggest that adsorption onto ferrihydrite mainly limits dissolved forms of these metals at circum-neutral pH, but with adsorption still occurring at the very high pH conditions predicted in our study; with up to 50 % sorption of As, Pb and V to ferrihydrite.

Although locations with high adsorption of As and V was predicted here, these metals were overall dominantly found in their dissolved inorganic fractions ( $\text{HAsO}_4^{2-}$  and  $\text{HVO}_4^{2-}$ ), indicating higher bioavailability of these compared to the others. As is well known to be highly dissolved at alkaline conditions (e.g. Early, 1999), especially in combination with non-humid climates (Smedley and Kinniburgh, 2002), which agrees with our results. Wällstedt et al. (2010) found, using the same modeling approach as this study, that both As and V were dominated by their dissolved fractions in streams with pH above 6.9. Further, since dissolved As concentrations reached above concentrations commonly found in freshwaters (Smedley and Kinniburgh, 2002) and health risk-based guideline values (WHO) in several locations, this metal is of additional toxic concern. A recent paper on As contamination in Mongolia (Pfeiffer et al., 2014) also

show high As concentrations in several regions, with highest values in areas of gold mines (including Zaamar Goldfield), highlighting this problem.

Model results also suggested that dissolved organic matter (DOC) influenced the speciation of several metals. DOC mainly controlled the dissolved form of Cd in both campaign years and Pb in 2013 and also impacted Zn. Many studies (e.g. Christensen et al., 1996; Hagedorn et al., 2000; Sauvé et al., 2000) have suggested DOC to be an important controlling factor for metal transport and fate in natural systems. A modeling approach by Weng et al. (2002) showed that metal complexes with organic matter could lead to dissolved metal concentrations of up to two orders of magnitude greater than without DOC, especially at alkaline conditions. This was most pronounced for Pb and Cu, whereas for Cd and Zn the effects were smaller. Antoniadis and Alloway (2002) however found that complexation with DOC significantly increased the dissolved concentrations of Cd and Zn; a result that agrees with the model results from this study. Additionally, binding of Fe and Al to organic matter has been shown to affect the binding of other metals as well. High concentrations of Fe and Al were shown by Tipping et al. (2002) to dominate the space at the organic material, thus increasing the dissolved concentrations of other metals which could no longer stay in a complex with the organic material. This was seen for Fe in our modeled system; with the scenario of increased DOC concentrations by 50 %, Fe-organic complexation also increased. This was also predicted to increase the dissolved fraction of previously adsorbed metals to ferrihydrite. If site specific DOC concentrations were higher than the here assumed values for the 2012 campaign, more metal-organic complexes could exist in our system than what was predicted, suggesting that the systematic under-prediction of the dissolved fraction for many metals in 2012 (Fig. 4a) could be related to DOC. Several studies (e.g. Schelker et al., 2012; Raymond and Saiers, 2010) have showed DOC fluctuations with up to several 100 %, for instance due to land use changes (deforestation) and storm events. Since data on DOC variability in the studied region is scarce (Yoshioka et al., 2002), it is possible that real DOC concentrations in 2012 were much higher than in 2013.

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As mentioned in the result Sect. 4.3 model and measured results on the speciation of Cr, Cu, Mn and Mo did not agree. A possible reason to this is the impact of adsorption to precipitates not included in the model. Apatites, a group of phosphate minerals, have commonly been suggested to be main solids limiting the dissolved form of several metals in natural systems (e.g. Chen et al., 1997). Notably, Hydroxyapatite, which is a natural calcium phosphate mineral within the apatite group, could be an important solid providing adsorption sites, since it was oversaturated (i.e. would precipitate if added as a possible solid phase) according to the model throughout the modeled river reach. Measured concentrations of particulate calcium and phosphorous, which are chemical constituents of the mineral, were also considerable in the Tuul River, suggesting that the mineral can be present in the system. However it would be premature to add this mineral to the model as a possible solid for adsorption, due to the current lack of an adsorption database for this phase. Nevertheless, several previous studies (e.g. Corami et al., 2008, 2012; Chen et al., 2010; Feng et al., 2010) have pointed to the importance of this solid for adsorption of metals, including Cr, Cu and Mo. For example, Cr adsorption to hydroxyapatite was shown experimentally by Asgari et al. (2012) to remove high fractions of dissolved Cr, up to pH 11. Hence, Hydroxyapatite could be an active control of several metals in non-acidic systems, even if its sorption database is currently lacking.

Linking the results from the speciation modelling with the measured concentrations and mass flow quantifications highlights the strong correlation between the predicted forms of precipitated Fe and Al and their high impact of suspended to total transport. The potential capacity of sediments to hold much more metals than corresponding volume of water is well known (e.g. Horowitz, 1991) and the combined effects of the mining activities; stirring up metal rich bottom sediments, and the non-acidic environment; precipitating Al and Fe and enhancing adsorption of several other metals, leads to overall high export of several metals in suspension from the site. This is in line with previous site assessments (Thorslund et al., 2012), suggesting a net contribution of metal loads transported from Zaamar Goldfield compared to natural mass flows. Since

concentrations of metals did not vary as much as discharges, discharge would be the main factor controlling the magnitude of mass flows. This impact of discharge on pollutant loads was also seen by Basu et al. (2010).

Considering combined influences of land use alterations (expansion of mining areas) and projected climate change (increasing the frequency of peak flow events; Altansukh and Davaa, 2011), which likely result in increased discharge and erosion in the Lake Baikal Drainage Basin, one can expect that the transport of metals from the site to the connected river system will also increase. The transport of metals to downstream regions, where geochemical conditions might be different (e.g. Yoshioka et al., 2002), could potentially increase the solubility and bioavailability of metals in suspension. This highlights the need for further studies focusing on large-scale implications of coupled hydrodynamic (governing the magnitude of suspended sediment transport) and geochemical (governing the mobility of metals between the sediment and water phase) processes.

## 6 Conclusions

Under non-acidic conditions, riverine suspended mass flows typically constitutes a dominating part of total mass flows, such as observed during our snapshot measurements in the Tuul river at Zaamar Goldfield. Total mass flows showed net increases over the mining site, reflecting export of several metals (Al, Cd, Fe, Mn, Pb and V) from the site to the downstream river system. Our results show that Zn partitioning had a particularly high sensitivity to pH under circum-neutral and high-pH conditions (pH 6 to 9.6), which contrasts to near 100% Zn dissolution under different conceivable acid mine drainage conditions (below pH 6). At site specific pH (8–9.6), between 50–80% of the available Zn was sorbed onto gibbsite.

Another main difference from acid mine drainage geochemistry is that the prevailing high pH causes precipitation of ferrihydrite and gibbsite, which removes between 90 to 100% of Fe and Al from solution. This notably influences the behavior of As, Pb and V

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since their solubilities are controlled by sorption onto ferrihydrite. For instance, 40 % of available Pb, 40 % of available V and 10 % of available As was sorbed to ferrihydrite at pH above 8. Despite high pH conditions, As and V predominantly occurred in dissolved inorganic forms ( $\text{HAsO}_4^{2-}$  and  $\text{HVO}_4^{2-}$ , respectively). Arsenic also showed dissolved concentrations above health risk-based guideline values in several locations and can thus be of main toxic concern in the upper Lake Baikal Drainage Basin.

Model results also indicated that the speciation of Cd, Pb and Zn was to large extent influenced by complexation with DOC (up to 70 % bound to DOC). In high pH environments, most metals have relatively low solubility in absence of DOC. However, present modelling showed that in particular the solubility Fe, Pb and Zn can increase considerably as DOC concentrations increase (up to 10 % higher solubility for 50 % DOC concentration increase). In high pH systems, seasonality of DOC concentrations (which could vary by several 100 %) can therefore have a major influence on the spreading and toxicity of these metals. Furthermore, the spreading of Fe, Pb and Zn can then also change considerably in response to climate and land use changes that impact DOC concentrations, such as increasing frequency of storm events, agricultural expansion and deforestation.

For most metals, model results on their dissolution generally agreed well (within 20 % difference in dissolved fractions) with observations in the Tuul River. However, the model predicted Cr, Cu and Mo to be 100 % in solution whereas observational data showed that up to 60 % of their mass could be attached to particles in suspension. A possible reason for this discrepancy is that the model neglected the influence of sorption to hydroxyapatite, an apatite mineral known to have high sorption capacity. Nevertheless, its adsorption database is currently lacking, which hinders appropriate sorption quantification. The potential importance of this mineral for metal partitioning under non-acidic conditions in general, and for the investigated Tuul River in particular, is emphasized by the fact that this mineral's constituents were observed on-site, with conditions being favorable for its precipitation.

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*Author contribution.* J. Thorslund did model simulations and manuscript preparation, with contributions from all co-authors. J. Thorslund, J. Jarsjö and S. Chalov developed research ideas, including field preparations, measurements and modelling approach. T. Wällstedt was responsible for modelling aspects, including parameter settings. M. Mörth was responsible for analytical methods, sampling handling and analysis. M. Lychagin contributed to geochemical interpretation.

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Sample	Date	Location	Longitude (decimal degrees)	Latitude
T5	24 Jun 2012 14 Sep 2013	Tuul River, upstream of Zaamar site	104.523	48.014
T5a	24 Jun 2012 14 Sep 2013	Tuul River, start of Zaamar site	104.306	48.158
T5b	23 Jun 2012 12 Sep 2013	Tuul River, at Zaamar site	104.325	48.229
T6	23 Jun 2012 11 Sep 2013	Tuul River, at Zaamar site	104.420	48.332
T6a	23 Jun 2012 11 Sep 2013	Tuul River, downstream of Zaamar site	104.512	48.389

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**Table 2.** Chosen oxidation states of the various components for the input file in Visual Minteq. Since samples were collected from river water, well oxygenated conditions were assumed and thus oxidized forms of the elements were chosen. Some elements (Al, Ca, K, Mg, Na, Ni, P, Pb, Si, Zn) only had one state in the program and could thus not be altered.

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Al<sup>3+</sup>, As(V), Ca<sup>2+</sup>, Cd<sup>2+</sup>, Co<sup>2+</sup>, Cr(VI), Cu<sup>2+</sup>, Fe<sup>3+</sup>, K<sup>+</sup>, Mg<sup>2+</sup>, Mn<sup>2+</sup>, Mo(VI),  
Na<sup>+</sup>, Ni<sup>2+</sup>, P (PO<sub>4</sub>), Pb<sup>2+</sup>, S (SO<sub>4</sub>), Cl<sup>-</sup>, Si (H<sub>4</sub>SiO<sub>4</sub>), V(V), Zn<sup>2+</sup>

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**Table 3.** Reaction and reaction constants of the possible solid phases allowed to precipitate if their solubility constant ( $\log K_s$ ) is exceeded (equations from Visual Minteq).

Mineral	Reaction	$\log K_s$ (25 °C)
Aluminium hydroxide (soil)	$\text{Al}(\text{OH})_3 + 3\text{H}^+ = \text{Al}^{3+} + 3\text{H}_2\text{O}$	8.29
Calcite	$\text{CaCO}_3 + 2\text{H}^+ = \text{Ca}^{2+} + \text{CO}_2(\text{g}) + \text{H}_2\text{O}$	-8.48
Ferrihydrite (aged)	$\text{Fe}(\text{OH})_3 + 3\text{H}^+ = \text{Fe}^{3+} + 3\text{H}_2\text{O}$	2.69
Gibbsite (C)	$\text{Al}(\text{OH})_3 + 3\text{H}^+ = \text{Al}^{3+} + 3\text{H}_2\text{O}$	7.74
Manganite	$\text{MnOOH} + 3\text{H}^+ = \text{Mn}^{3+} + 2\text{H}_2\text{O}$	25.3

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**Table 4.** Measured geochemical parameters in the five water samples along the Tuul River reach (T5, T5a, T5b, T6, T6a) in 2012.

Site/Parameter	T5 2012/2013	T5a 2012/2013	T5b 2012/2013	T6 2012/2013	T6a 2012/2013
pH	9.2/9.2	9.6/7.6	8.8/8.0	8.5/8.8	8.7/8.5
Temp (°C)	16.9/19.5	19.2/17.0	22.4/17.0	19.1/17.0	19.1/15.6
pe <sup>a</sup>	11.9/12.1	11.0/13.0	11.8/12.6	12.1/11.8	12.3/12.1
Alkalinity (mg L <sup>-1</sup> )	125/520	100/520	98.0/310	93.1 <sup>b</sup> /310	80.0 <sup>b</sup> /310
DOC (mg L <sup>-1</sup> ) <sup>b</sup>	13.4/11.7	5.77/11.2	7.20/10.7	6.43/11.5	4.73/11.7

<sup>a</sup> calculated according to Eq. (4), <sup>b</sup> reported data from Lee et al. (2006)

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**Table A1.** Detection limits (DL) for each metal from the inductive coupled plasma optical emission spectrometry (ICP-OES) analysis.

Metal DL ( $\mu\text{g L}^{-1}$ )	2012	2013
As	9.76	9.76
Cd	0.015	0.015
Cr	0.10	0.04
Cu	0.20	0.01
Fe	0.07	0.07
Mn	0.018	0.018
Mo	2.29	2.29
Pb	0.10	0.10
V	0.085	0.085
Zn	0.028	0.028

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**Table B1.** Maximum permissible concentrations levels ( $\mu\text{g L}^{-1}$ ) in drinking water according to the World Health Organization (WHO, 2006) and the Mongolian State Standard (AATA, 2008).

	Al	As	Cd	Cr	Cu	Fe	Mn	Mo	Pb	V	Zn
WHO	200	10	3.00	50	2000	300	400	70	10	200	3000
Mongolian	500	10	–	–	–	300	100	–	10	–	5000

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	Al	As	Cd	Cr	Cu	Fe	Mn	Mo	Pb	V	Zn
% Dissolved measured											
T5	0.3	96.5	NA	NA	67.5	3.5	6.	84.5	72.3	88.7	57.4
T5a	0.3	93.4	NA	10.	26.1	2.4	2.	33.5	NA	73.1	24.7
T5b	3.6	85.7	87.1	39.4	NA	3.7	89.1	88.9	NA	78.	28.1
T6	1.1	NA	67.5	21.2	NA	2.3	2.6	41.2	12.1	62.8	44.5
T6a	4.	92.2	86.9	35.1	NA	2.7	4.5	80.5	NA	82.2	64.4
% Dissolved modelled											
T5	2.63	72.3	NA	NA	100	2.54	0.00	100	48.4	55.0	45.3
T5a	1.77	39.8	NA	100	99.8	0.21	0.00	100	NA	37.1	5.76
T5b	1.29	94.5	90.2	100	NA	0.82	0.00	100	NA	75.6	23.5
T6	0.05	NA	53.0	100	NA	0.46	0.00	100	19.4	52.8	3.08
T6a	0.78	98.6	91.4	100	NA	2.03	0.00	100	NA	89.2	24.2

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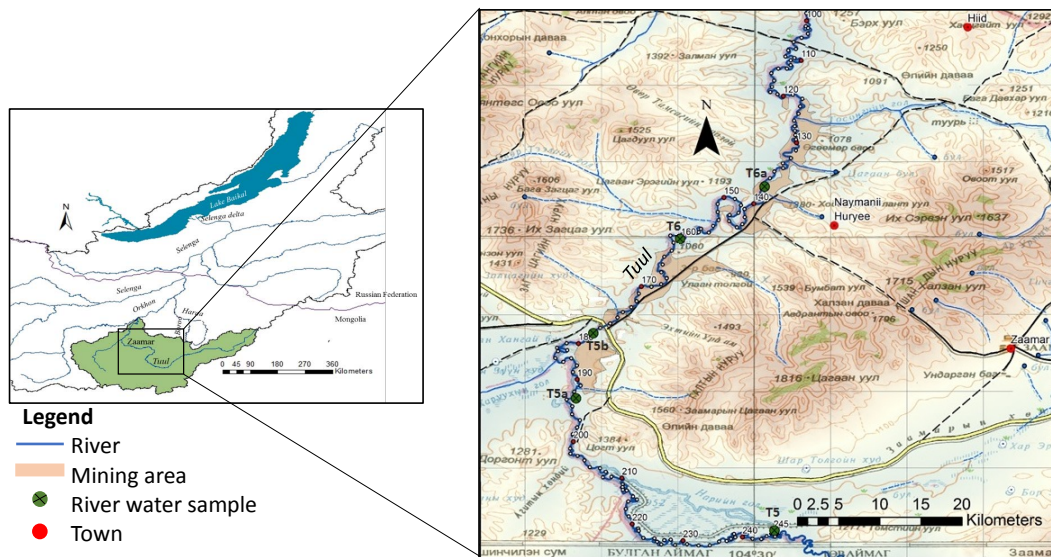
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**Table C2.** Comparison of measured and modeled dissolved fractions of the selected metals in 2013.

	Al	As	Cd	Cr	Cu	Fe	Mn	Mo	Pb	V	Zn
% Dissolved Measured											
T5	2.49	NA	NA	66.8	55.2	8.8	10.6	NA	53.4	40.7	70.2
T5a	2.55	NA	45.1	92.8	83.7	6.7	42.3	NA	57.2	64.9	48.3
T5b	6.07	NA	NA	74.8	69.9	7.5	25.1	82.0	50.9	57.5	105
T6	0.92	76.4	76	102	62.3	2.2	15.6	77.4	40.7	45.0	31.0
T6a	11.9	NA	NA	75.4	83.6	3.7	11.7	NA	37.8	45.8	48.7
% Dissolved Modelled											
T5	1.54	NA	NA	100	100	4.27	0.00	NA	71.0	63.9	71.9
T5a	0.60	NA	97.0	100	100	7.23	0.00	NA	67.0	56.6	75.0
T5b	0.29	NA	NA	100	100	2.66	0.00	100	58.0	49.1	51.0
T6	0.63	85.0	91.0	100	100	1.21	0.00	100	41.1	53.7	32.0
T6a	0.29	NA	NA	100	100	1.58	0.00	NA	45.2	49.7	21.0

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**Figure 1.** Map of the Baikal Basin with the Zaamar mining area (zoomed) located in the Mongolian part of the basin, along the Tuul River (author J. Thorslund).

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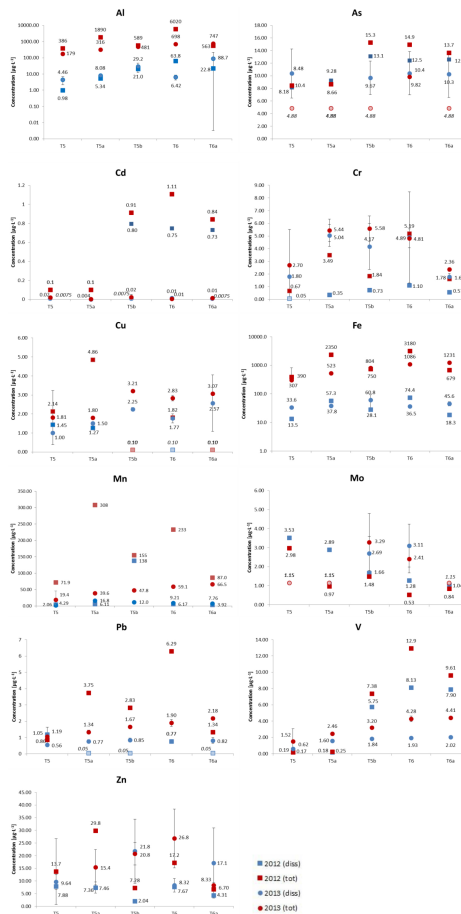
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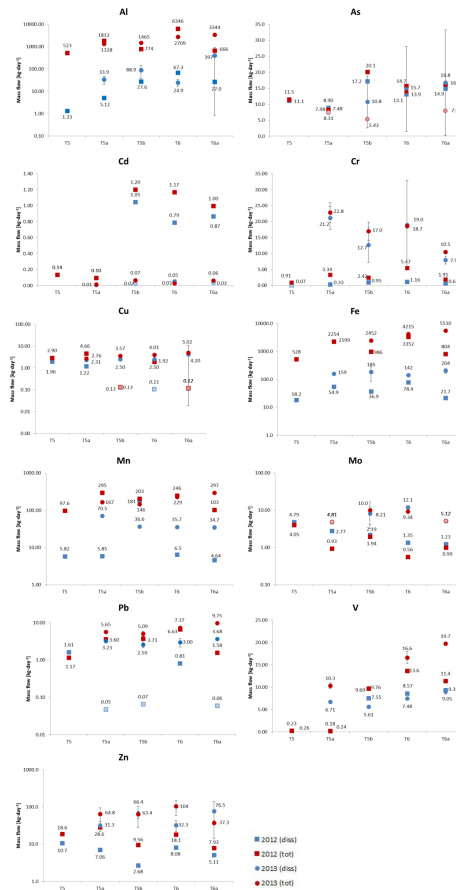
**Figure 2.** Dissolved (blue) and total (red) concentrations of Al, As, Cd, Cr, Cu, Fe, Mn, Mo, Pb, V and Zn at the five sampling locations along the Tuel River and the Zaamar Goldfield, from the sampling campaigns in 2012 (squares) and 2013 (circles).

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**Figure 3.** Estimated dissolved (blue) and total (red) mass flows, in kg per day, of Al, As, Cd, Cr, Cu, Fe, Mn, Mo, Pb, V and Zn at the five sampling locations (T5–T6a) along the Tuul River and the Zaamar Goldfield, from the sampling campaigns in 2012 (squares) and 2013 (circles).

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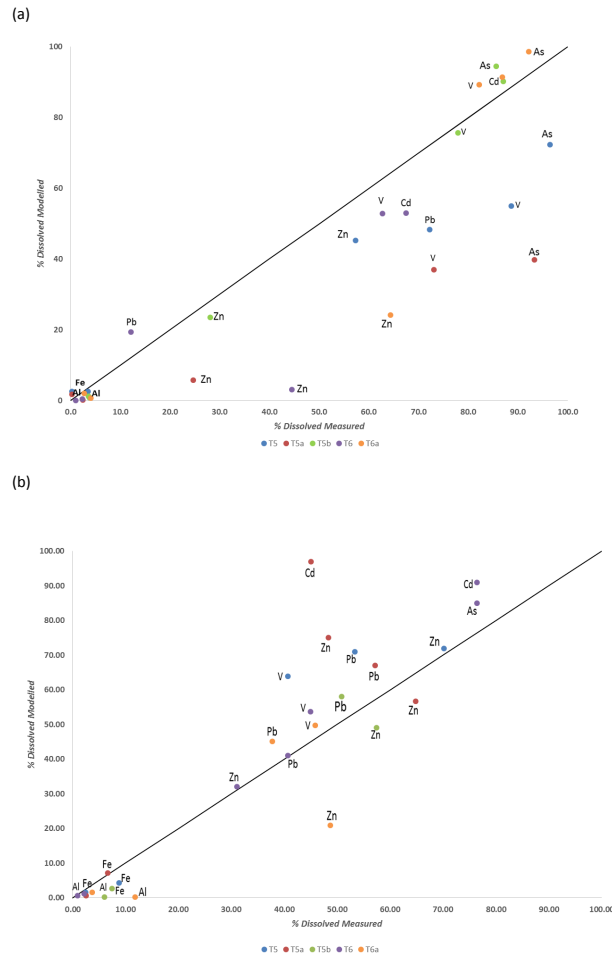


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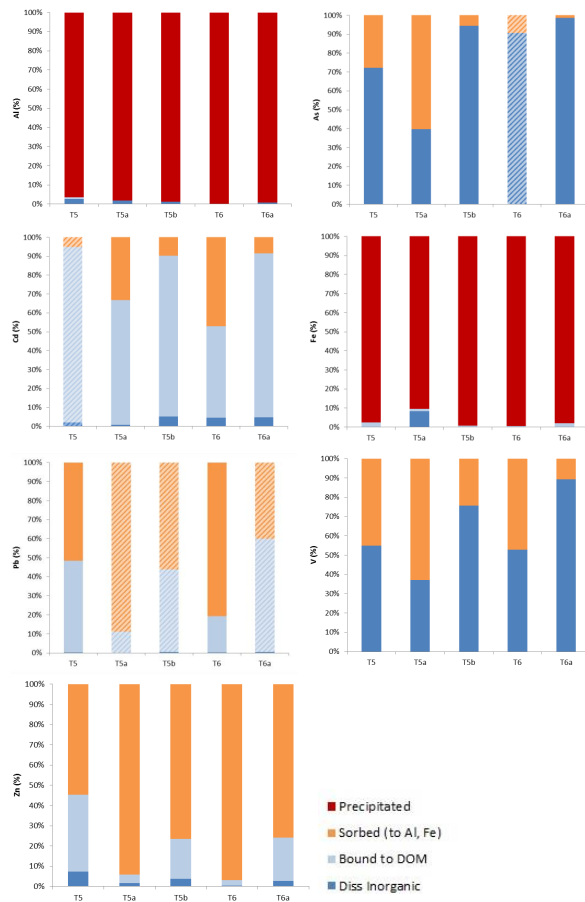
**Figure 4.** Comparison between measured and modeled results for the dissolved fractions (percentage) of metals in the **(a)** 2012 campaign and the **(b)** 2013 campaign.

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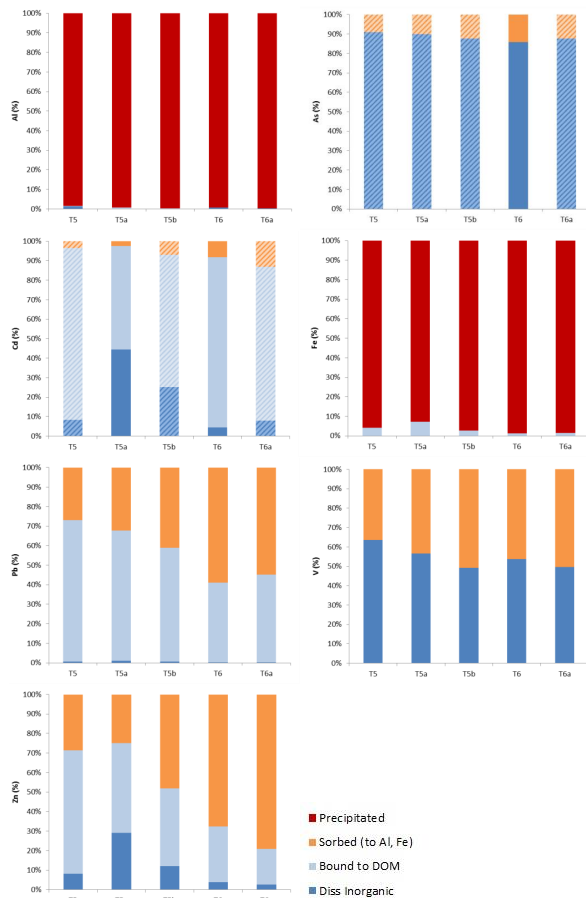


**Figure 5.** Predicted speciation of selected heavy metals between “dissolved inorganic”, “bound to DOM”, “sorbed” and “precipitated” in the five sampling locations along the Tuul River in the 2012 campaign. Dashed lines indicate model prediction from detection limit (DL) concentrations.

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**Figure 6.** Predicted speciation of selected heavy metals between “dissolved inorganic”, “bound to DOM”, “sorbed” and “precipitated” in the five sampling locations along the Tuul River in the 2013 campaign. Dashed lines indicate model prediction from detection limit (DL) concentrations.