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ORIGINAL RESEARCH ARTICLE

Effects of landslide hazards on quality of stream water and sediments

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ABSTRACT

BACKGROUND AND OBJECTIVES: Landslide disasters in Thailand between 1970 and 2011 revealed a notable pattern: they primarily originated on mountain slopes, distinguished by a deeper soil profile. This soil profile comprised clay loam and sandy loam textures and was situated over aged geological formations of granite and shale rocks. The affected areas included the southern and northern provinces of Thailand.

This study investigated the consequences of landslide hazards on stream water and sediment quality in two watersheds: the Mae Phul–Mae Prong watershed in Uttaradit province, the northern part of Thailand, and the Klong Kram watershed in Surat Thani province, the southern part of Thailand. These watersheds had experienced recurrent landslides, primarily on mountain slopes characterized by deep clayey and sandy loam soils over old granite and shale rock types as well as old granite limestone.

METHODS: During wet and dry periods in April and November 2015, 108 samples were collected from 18 stations (9 stations in the Klong Kram watershed and 9 stations in the Mae Phul–Mae Prong watershed). These samples included upland soil, stream water, and sediments. For upland soils, 1 kilogram samples were collected through auger and V-shaped pit techniques using a stainless-steel spade, with composite sampling conducted at 0–30 centimeters across all 18 stations. Stream water was collected in one part using a 1-L polyethylene bottle at 30 centimeter from the stream layer, while other samples were compositely collected in sterilized glass bottles to determine coliforms. Soil and sediment samples were compositely collected from the bottom using a stainless-steel spade. All samples were stored at 4 degrees Celsius and transported to a laboratory for analysis. The insight gained from these collection efforts elucidated the dynamics of landslide impacts at the spatial scale for the two watersheds.

FINDINGS: Most water samples met Thai surface water quality standard for various parameters; however, microbial contamination of the water samples attributed to community activities along stream banks was detected. Notably, arsenic was consistently detected in upland soil, stream water, and sediment samples. For Uttaradit, the average arsenic concentrations were 0.22 ± 0.09 milligram per kilogram, 0.01 ± 0.14 milligram per liter, and 9.74 ± 4.42 milligram per kilogram in upland soil, water, and sediment samples, respectively. For Surat Thani, arsenic concentrations were 87.63 ± 208.83 milligram per kilogram, 0.01 ± 0.01 milligram per liter, and 19.44 ± 36.38 milligram per kilogram in upland soil, water, and sediment samples, respectively, particularly near landslide scars where the arsenic concentrations were significantly higher in sediments and upland soils compared with stream water, highlighting the role of landslides near streams. These data suggest that sediment transport from upland soil in the landslide scar into stream water affects water quality, particularly in terms of arsenic concentration near the landslide scar, often surpassing natural standards.

CONCLUSION: The study concluded that stream water was directly affected by landslides as these watersheds were unsuitable for consumption due to arsenic and microbial contaminations. This conclusion emphasizes the critical need to incorporate landslide hazard considerations into watershed management practices to safeguard downstream communities and preserve water resources.

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5

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2

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INTRODUCTION

Landslides are a process associated with mass transport deposits that involves the descent of soil masses along with litter, shrubs, and trees from steep slopes and highland areas, regardless of vegetation cover. Sedimentary geology describes these movements as slides, slumps, and debris flows (Shanmugam 2015). Landslides typically occur during monsoon owing to intense, heavy rainfall lasting an extended duration, sometimes exceeding 3 days with more than 400 mm of rainfall per storm; moreover, these events often result in elevating sand levels in rivers and impacting reservoir water quality and river ecohydrology (Opiso *et al.,* 2016; Chiang *et al.,* 2021). Researchers have reported a key finding that landslide detection probability increases with rainfall intensity and cumulative rainfall (Paolini *et al*., 2005). In tropical mountain areas, typhoon-induced rainstorms trigger extensive landslides and deliver large amounts of sediment to watersheds (Paolini *et al*., 2005; Chen, 2009; Lin and Chen, 2012). Fracture rocks (cracked rocks) would presumably be an important factor as a catalyst for the landslide process. Quaternary sediments and the occurrence of dense bedded basalt cover beds are responsible for landslide susceptibility at a critical slope angle of 3°–5 degrees°. A reduction in the movement-triggering rainfall threshold causes a decrease in the overall slope stability (Damm and Terhorst, 2010; Preuth *et al*., 2010). The above-ground forest cover may not seem to play an important role in accelerating the landslide process; however, the root system holds the soil particles underneath the mountain slope soil surface. Additionally, other natural disasters, such as earthquake and mudflows, have been observed to trigger landslides, reshaping the structure of rivers. These events can lead to sudden changes in river water quality, characterized by elevated levels of suspended solids (Payus *et al.,* 2023). Land use changes affect the landslide potential due to land cover changes causing topsoil detachment during rainfall (Rijsdijk *et al*., 2007). Gradual land use and land cover changes have been reported in several areas worldwide, e.g., the cultivated area declined from 37.3% to 20.7% in Yanchang City, China during 1990 to 2008 (Zhen *et al*., 2014). The substantial urban growth involving rural roads, trails, and settlements, which substantially produced runoff and sediment at the watershed scale (Trincsi *et al*., 2014). Accumulations of released ions are highly concentrated at the base of slopes. Geographically, these ions migrate along with water, influencing the ecohydrology of watersheds (Fan, 2017); these natural disasters are main contributors to sediment yield (La Licata *et al.,* 2023). In a previous study, root cohesion increased from 300 to 1500 Pa causing a two-third reduction in the number of landslide occurrences; moreover, deep-rooted vetiver grass demonstrated high water absorption that tended to accelerate the landslide process for a weathered rock slope of approximately 60^{o} and depth of 0.8 m (Bathurst *et al*., 2010). Furthermore, vetiver roots can potentially provide a pathway for water infiltration and increase pore water pressure, thus reducing the safety factor of a slope by approximately 10% (Jotisankasa *et al*., 2014). A landslide event contributes to sediment sequences in depositional basins such as lakes, swamps, estuarine areas, coastal wetlands, coastal foothills, and the nearshore and offshore zones of continental platforms (Glade, 2003; Sewell *et al*., 2015). Debris and rock piles in the catchment area of a river after a landslide contaminate stream water with heavy metals and sediment, causing flash floods and adversely affecting aquatic habitats and water quality (Turner *et al*., 2010; Chen, 2009). Notably, contamination caused by heavy metals, particularly arsenic (As), cadmium (Cd), and copper (Cu), along with their coloration with total nitrogen (TN) and phosphorus, originating from upland parent materials, is evident during these events. Thailand has faced notable challenges related to landslide and mudflow disasters since the 1970s, resulting in substantial loss of human and animal lives and property damage. The main causes of landslide events in the country include increased rainfall, steep slopes owing to changes in land use patterns (i.e., cultivation practices), soil fertility enhancement, and effective management of landslide and water resources (Priyono *et al.,* 2022). The estimated cost of the damages resulting from these events was more than USD 8.4 million. As part of prevention efforts, the Thai government selected Surat Thani and Uttaradit Provinces as representative sites from the south and north of the country, respectively. The focus of the associated study was to assess the influence of landslide hazards on potential stream water pollution, a critical concern with respect to human consumption. Data from this study are crucial for preserving water quality in landslide-prone regions, thereby ensuring the safety and health of affected communities. Addressing landslide hazards and their impact on stream water quality emerges as a pivotal element of disaster management and environmental protection in Thailand. To achieve these objectives, in this study, upland soil, sediment, and water samples affected by landslide events were collected. This study aims to investigate the consequences of landslide hazards on stream water and sediment quality in two important watersheds in Thailand, namely, the Mae Phul–Mae Prong (UT) watershed in Uttaradit Province and the Klong Kram (SR) watershed in Surat Thani province, during the wet and dry periods in April 2015 and November 2015, respectively.

MATERIALS AND METHODS

Study areas

Surat Thani and Uttaradit Provinces were selected as the study sites from the south and north of Thailand, respectively, considering their previous history with severe landslide disasters. The study site in the north was in Mae Phul sub-district, Lab Lae district, Uttaradit Province (Universal Transverse Mercator 47Q 598067-610873 East (E), 1970074-1954242 North(N)) The study site in the south of Thailand was in Ban Na Tham, Tha Utae sub-district, Karnchanadit district, Surat Thani province (Universal Transverse Mercator 47P 575370-580638E, 1000957-0991428 N) (Fig. 1).

Sampling stations

Samples of upland soil, soil sediment, and stream water were collected from nine stations in the SR watershed and nine stations in the UT watershed (total: 18 stations) during the wet in April, 2015 and dry in November, 2015 periods. Three samples each of upland soil, soil sediment, and stream water were collected from each station; thus, the total number of samples for upland soil, soil sediment, and stream water was 108 samples totaling at 324 samples for all samples was collected (Fig. 2).

Upland soil sampling

One hundred and eight soil samples were collected using an auger and a V-shaped pit with a stainless-steel spade, and composite sampling methods were used at upland soil sampling stations. Soil samples were collected from 18 stations (UT watershed: 9 stations and SR watershed: 9 stations) at a depth of 0–30 centimeter (cm) as this was the average depth for plant roots. Three sediment samples were collected at each station during the dry and wet periods. Each soil sample weighed 1 kilogram (kg).

Sediment sampling

Sediment samples were collected using a stainless-steel spade using composite sampling from 18 stations, including 9 stations in the UT watershed and 9 stations in the SR watershed. At each station, three sediment samples were collected during the wet and dry periods, resulting in a total of 108 sediment samples. The samples were kept in individual black bags to prevent photodecomposition and stored at 4°C during transport to the laboratory.

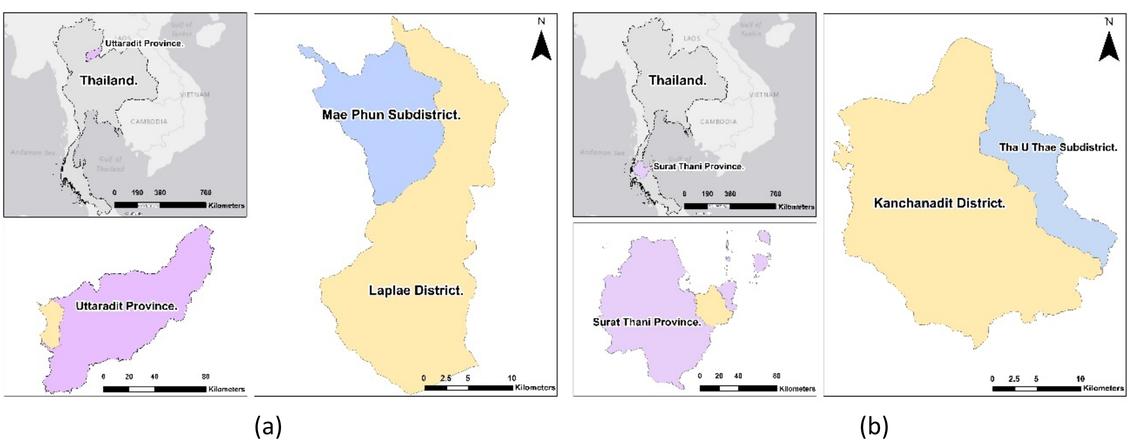
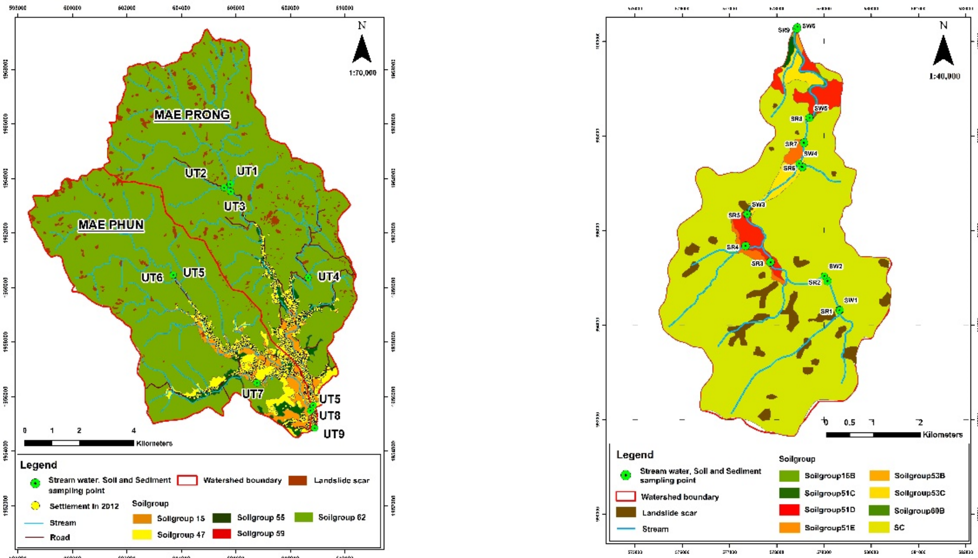


Fig. 1: Geographic location of the study area in (a) Uttaradit Province and (b) Surat Thani province, Thailand



(a) (b)

Fig. 2: Sampling stations for upland soil, soil sediment, and stream water in a) the Mae Phul–Mae Prong watershed, Mae Phul sub-district, Lablae district, Uttaradit Province, and northern Thailand; b) the Klong Kram watershed, Tha Utae sub-district, Kanchanadit district, Surat Thani province, and southern Thailand

Stream water sampling and analysis

Four-liter samples were collected from 18 stations in the wet and dry periods. One part of the stream water sample was collected using a 1-L polyethylene bottle at 30 cm from the top of the stream layer. The other part of the sample was collected separately in a sterilized glass bottle for the determination of coliforms. All samples were stored at 4 degrees Celsius (°C) and transported directly to the laboratory for analysis. The laboratory procedures included the measurement of various parameters, such as potential Hydrogen (pH), electrical conductivity (EC), total solids (TS), total dissolved solids (TDS), temperature, turbidity, ammonia, nitrate (NO_3^-), phosphate (PO_4^-), biochemical oxygen demand (BOD), dissolved oxygen (DO), iron (Fe), manganese (Mn), arsenic (As), lead (Pb), calcium (Ca) and magnesium (Mg), following standard methods outlined in APHA (2017), as detailed in Table 1. Notably, dissolved oxygen (DO) measurements were conducted in the field (APHA, 2017). Specifically, heavy-metal analysis of water samples was conducted using flame atomic absorption (FAA) spectrophotometry (FAAS). A 100 milliliter (mL) water sample was mixed with 10 mL of concentrated nitric acid in an Erlenmeyer flask and gently heated until reaching a slow boil, maintaining a final volume of approximately 10 mL. After cooling, the sample was transferred to a 100-mL volumetric flask and adjusted with respect to volume using distilled water, followed by filtration through Whatman No. 42 filter paper to eliminate any impurities. Subsequently, the filtered sample was subjected to heavy-metal content analysis using FAAS as per APHA guidelines (2017). Analysis to determine the quantity of bacteria, including coliform and *Escherichia coli* (*E. coli)*, was conducted to investigate the contamination caused by fecal bacteria for assessing the wastewater discharge activity of the community into the stream. Total coliform bacteria (TCB), fecal coliform bacteria (FCB), and *E. coli* were analyzed in the laboratory using the multiple-tube fermentation method: APHA-9221, following APHA standards (2017). Bacteria were solely collected and analyzed as part of the water quality assessment; this procedure was not conducted in case of the sediment and upland soil samples because this study focused on impact of human activities on the river. Coliform bacteria are aerobic, while sediments, particularly in landslide-prone areas, show limited oxygen content, promoting anaerobic conditions (Kondratyeva *et al.,* 2020).

Table 1: Standard methods for water and wastewater analysis

Parameters	Methods	Sources of methods
Temperature	Thermometer	APHA, 2017: 23rd edition, part 2550
TS	Total solids dried at 103–105°C	APHA, 2017: 23rd edition, part 2540 B
pH	pH meter	APHA, 2017: 23rd edition, part 4500-H^+ C
DO	Azide modification	APHA, 2017: 23rd edition
BOD	Azide modification	APHA, 2017: 23rd edition, part 5210B
NH_3 N_2	Phenate method	APHA, 2017: 23rd edition, part 4500-NH_3 C
NO_3^- N_2	Cadmium reduction	
PO_4^{3-}	Vanadomolybdo-phosphoric acid colorimetric method	APHA, 2017: 23rd edition, part 4500-P
Ca	FAAS	APHA, 2017: 23rd edition, part 3500
Mg	FAAS	APHA, 2017: 23rd edition, part 3500
Fe	FAAS	APHA, 2017: 23rd edition, part 3500
Mn	FAAS	APHA, 2017: 23rd edition, part 3500
Zn	FAAS	APHA, 2017: 23rd edition, part 3500
Pb	FAAS	APHA, 2017: 23rd edition, part 3500
As	AAS (vapor generation technique)	APHA, 2017: 23rd edition, part 3500
Cd	FAAS	APHA, 2017: 23rd edition, part 3500
Hg	AAS (vapor generation technique)	APHA, 2017: 23rd edition, part 3500
TCB	MPN method	APHA, 2017: 23rd edition, part 9221 B
FCB	MPN method	APHA, 2017: 23rd edition, part 9221 B
E.coli	Presence-absence test	APHA, 2017: 23rd edition, part 9221 B

Table 2: Methods for soil and sediment analysis

Parameters	Methods	Sources of methods
Soil texture	Hydrometer method	Okalebo *et al.*, 2002
pH	pH meter	Okalebo *et al.*, 2002
Ca	Atomic absorption spectrophotometry	Okalebo *et al.*, 2002
Mg	Atomic absorption spectrophotometry	Okalebo *et al.*, 2002
As	Acid digestion (USEPA Method 3050B)	USEPA, 1996
Fe	Acid digestion (USEPA Method 3050B)	USEPA, 1996
Mn	Acid digestion (USEPA Method 3050B)	USEPA, 1996
Pb	Acid digestion (USEPA Method 3050B)	USEPA, 1996

Soil and sediment analysis

The soil and sediment samples were air-dried under shade conditions, subsequently finely ground, and finally sieved through sieves having different sizes: 2 and 0.5 mm. The sieved samples were analyzed to determine soil characteristics, including physical attributes such as texture and an array of chemical properties, including pH levels and the presence of heavy metals, specifically As, Fe, Mn, and Pb (Okalebo *et al.,* 2002). Heavy metals from the soil matrix and sediment samples were dissolved using USEPA Method 3050B, which involves the use of acid digestion solutions comprising 30% nitric acid and 70% hydrochloric acid. The acid solutions were added to the soil samples, followed by a controlled heating process to facilitate metal dissolution. Thereafter, the samples were cooled and filtered to eliminate solid residues. Heavy-metal concentrations were measured using an FAA spectrophotometer (model: Agilent 240FSAA; serial No. MY14150004) (USEPA, 1996), as shown in Table 2.

RESULTS AND DISCUSSION

Upland soil characteristics in the landslide areas

The results for the upland soil samples analyzed for the percentage of soil content, organic matter, and heavy metals are shown in Fig. 3. The percentages of sand, silt, and clay in the SR watershed were higher than in the UT watershed. The percentage of sand was higher than those of clay and silt. The soil was slightly acidic, with average pH values of 5.2 ± 0.82 and 4.81

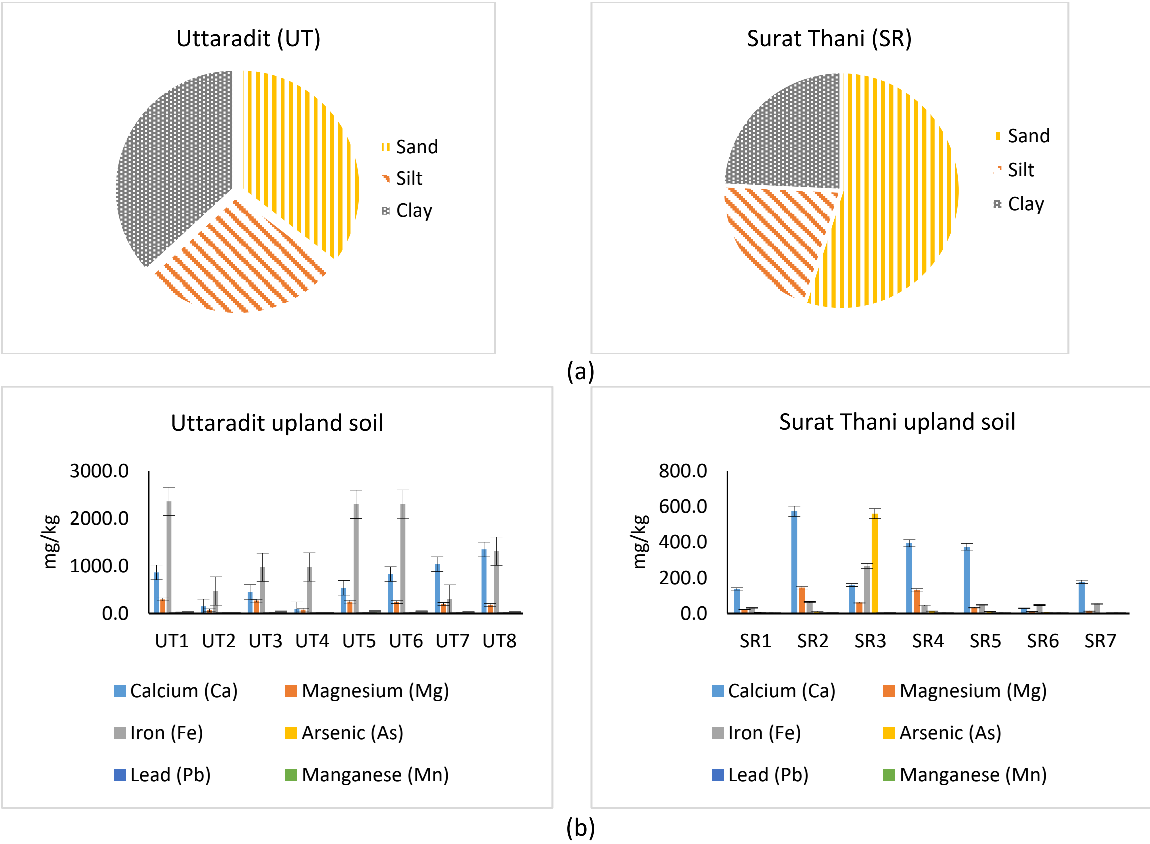


Fig. 3: Upland soil quality a) Soil texture; b) Heavy metals

± 0.42 in the UT and SR watersheds, respectively. In the SR watershed, Ca and Mg concentrations were considerably high owing to the parent material being limestone. Furthermore, As concentration was considerably high in the landslide scar area (station No. 2 in the SR watershed) because this area was geologically composed of granite and limestone. In the UT watershed, Pb, Mn, and Fe concentrations were high owing to the parent material being mudstone or shale. Additionally, the As concentration in the SR watershed exceeded the standard limits. The aforementioned results indicates that the soil characteristics were related to the parent material or the rock-weathering process in the slip zone soil, which was silty clay. The slip zone soil was formed from mudstone, and the main mineral components of the slip zone were montmorillonite, illite, feldspar, and quartz. The soil in the landslide impact area had become compacted and dry. (Cheng *et al*., 2012; Jian *et al*., 2019). The Pearson correlation analysis results showed a significant correlation between erodibility indices and certain soil properties such as clay and the sand fraction of soils ($p < 0.05$, $p < 0.01$). Soils were weakened by land use activity, and their chemical properties and bulk density were low; moreover, soil particles were not consolidated (Karkanc *et al*., 2008). Therefore, they detached easily when impacted by flood water.

Furthermore, the texture of upland soil in both the watersheds remained consistent between 2015 and 2022, maintaining clay loam in the UT watershed and sandy clay loam in the SR watershed. In line with this result, upland soil parameters, including Mg, Mn, Fe, Pb, and As concentrations, showed no significant differences ($p > 0.05$) between 2015 and 2022. However, a decline in Ca concentration was observed in both the watersheds (from 666.20 ± 408.38 to 264.52 ± 157.65 milligram per kilogram (mg/kg) and 264.68 ± 189.29 to 157 ± 104.21 mg/kg in the UT and SR watersheds, respectively). This decrease can be attributed to the leaching process of Ca induced by rainfall infiltrating the soil, soil erosion, and runoff processes. Heavy rainfall has the potential to trigger erosion and landslides on hillslopes, impacting stream water quality (Chiang *et al.*, 2021). Contaminated runoff from landslides could release pollutants, elevating toxicity levels in nearby freshwater for an extended period, possibly spanning several years. The natural erosion of the entire deposit might take more than 60 years. The positive correlation observed between the ratio of the landslide area to the catchment area and calcium ion (Ca^{2+}) and bicarbonate (HCO_3^-) levels in stream water indicates interactions with sliding surfaces and landslide deposits. Shallow landslides posed risks to both soil movement and water quality (Göransson *et al*., 2018; Yoshihara *et al*., 2022).

Sediment in a landslide area

The results are shown in Fig. 4. The sediments in the SR watershed mostly contained sand particles (>80%) in both the watersheds; moreover, these sediments contained negligible amounts of silt and clay particles. The results indicated an upland soil wherein sand particles were common. The pH of upland soil indicated that the pH of the soil sediment was neutral owing to a buffer reaction in the soil;; ; accordingly, the average pH values of the soil sediments in the SR and UT watersheds were 7.51 ± 0.37 and 6.84 ± 0.56, respectively. Sediment in streams following landslide events leads to their deposits in stream water. This finding was also reported in previous studies such as the shallow landslide and river bed variations associated with extreme rainfall-runoff events in a granitoid, mountainous, forested watershed in Japan, resulting in shallow landslides (<1.0 m depth) based on physical processes (Mouri *et al*., 2011). Therefore, rainfall and stormwater delivered large amounts of colluvial sediment into the river, leading to a twofold increase in the discharge of post-seismic sediment; however, the precipitation was only half that of the pre-earthquake rate (Chuang *et al*., 2009). The sediment textures of clays in the UT and SR watersheds showed a slight difference, with higher clay content observed in UT compared with SR (6.69% ± 3.50% to 1.91% ± 0.81%) during the dry and wet periods. This difference could be attributed to the parent materials of UT, which comprised mudstone, while SR comprised limestone and granite. The sediment analysis revealed no significant differences in Pb concentrations between the two watersheds during the dry and wet periods ($p > 0.05$). For UT, the average Pb concentration was 35.39 mg/kg, while for SR, it was 15.43 mg/kg. However, Ca concentration in the clay of UT (188.66 mg/kg) was significantly higher than in the clay of SR (243.17 mg/kg; $p < 0.05$). This difference can be attributed to the shale rock parent materials of UT, known to contain higher Ca concentrations (Sanghapi *et al.*, 2016). Furthermore, As concentration in the sediment of SR (19.46 mg/kg) was higher than in the sediment of UT (9.74 mg/kg). This difference in heavy-metal concentrations can be attributed to the parent materials of the soil in the SR watershed, primarily comprising granite rock, known to have elevated levels of these heavy metals (Uchida *et al.,* 2017). Comparison of the soil texture, pH, and Ca, Mg, Fe, Mn, Pb, and As concentrations of the clay between the UT and SR watersheds from 2015 to 2022 revealed considerable similarity, with no significant differences noted.

Stream water quality

Runoff processes after a landslide affect the stream water quality, particularly with respect to water consumption by residents near the landslide area. The SR and the UT watersheds experienced major landslide events in 2011 and 2006, respectively. Evidence of the considerable effects of these events on water quality is highlighted by the fact that water from the watersheds is used for consumption. Traces of heavy-metal elements were found in the sediment, particularly in landslide-prone areas, leading to their runoff into water. Stream water was collected for 108 samples in the watersheds equally in the wet and dry periods in 2014 as marginal seasonal variation was observed (Mark *et al.,* 2024). Results showed that most water quality indices were within the Thai standard limits for surface water quality, according to the (National Environment Board, 1994). As shown in Fig. 4, BOD values were higher for the UT watershed than for the SR watershed, although the station in proximity to residential areas in the SR watershed showed BOD values exceeding the standard limit (≤4.0 mg/L). However, the As concentrations at most locations in the wet and dry periods did not exceed the Thai surface water quality standards (<0.01

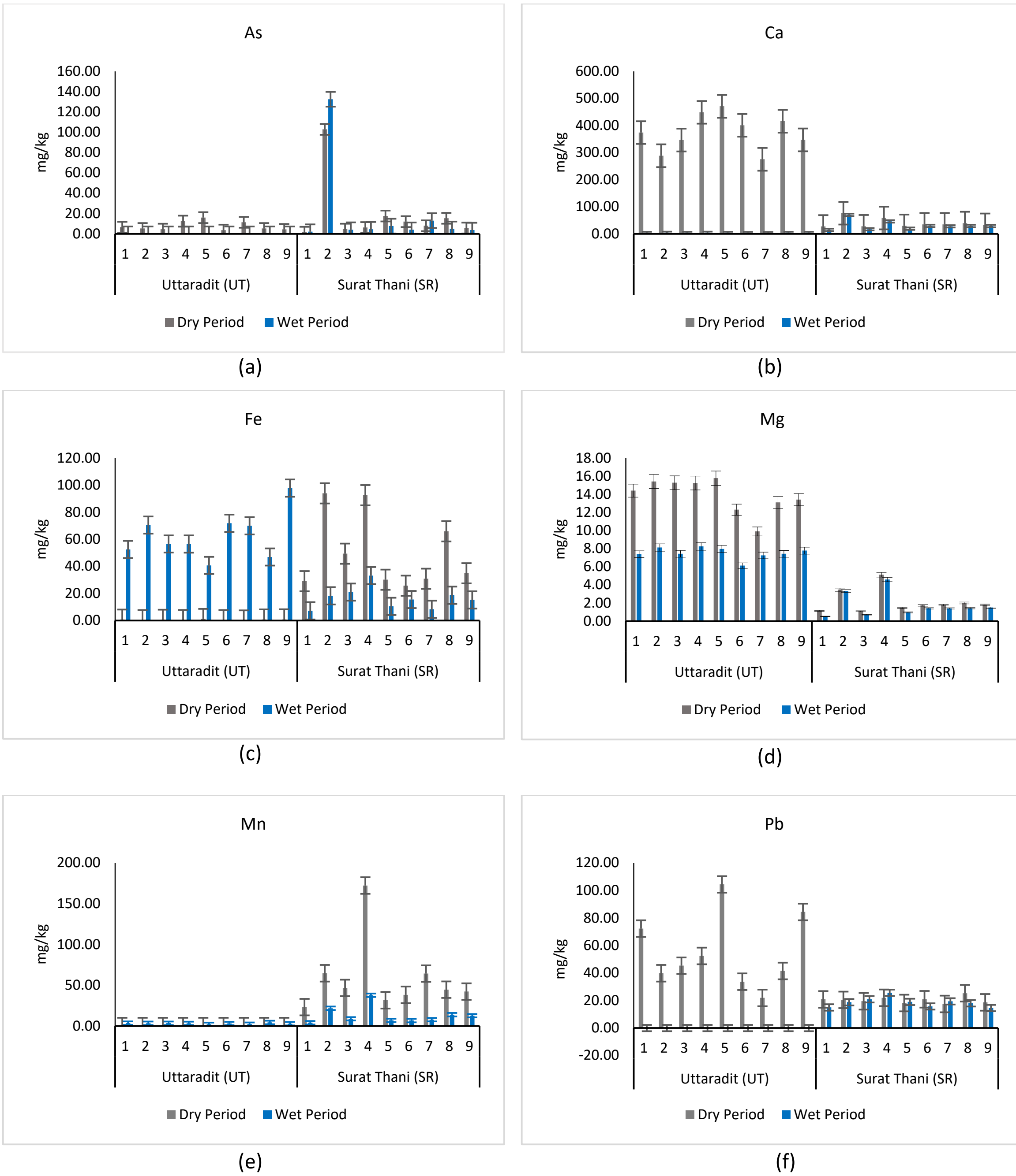


Fig. 4: Soil quality in Mae Phul–Mae Prong, Uttaradit and Klong Kram, Surat Thani during the wet and dry period. a) As; b) Cd c) Fe; d) Mg; e) Mn; f) Pb

mg/L), with the exception of station No. 2 in the SR watershed (0.02 mg/L), which was in proximity to the landslide scar. Furthermore, the total and fecal coliforms did not exceed the standards; however, at every sampling station, the *E. coli* levels exceeded the permissible standards for drinking water (National

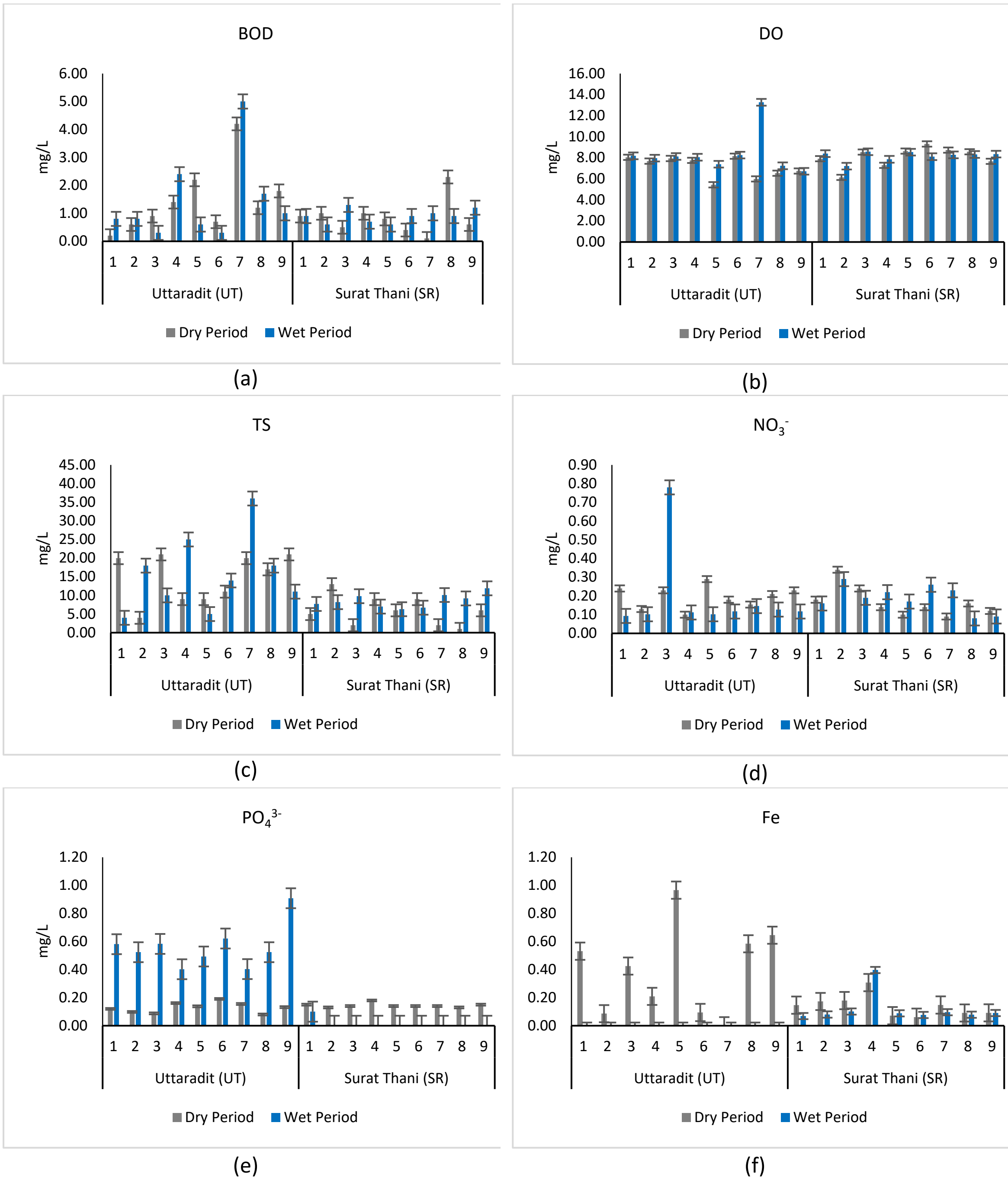
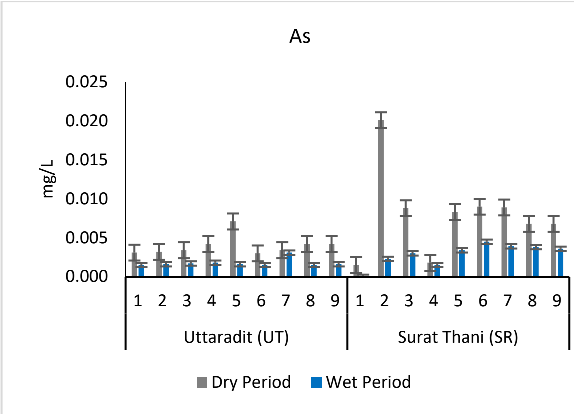


Fig. 5: Water quality in Mae Phul–Mae Prong, Uttaradit and Klong Kram, Surat Thani during the wet and dry period. a) BOD; b) DO; c) TS; d) NO^{3}-; e) PO_4^{3-}; f) Fe; g) As; h) Ca; i) Mg; j) Mn; k) Pb; l) Total coliform bacteria; m) FCB; n) *E. coli*



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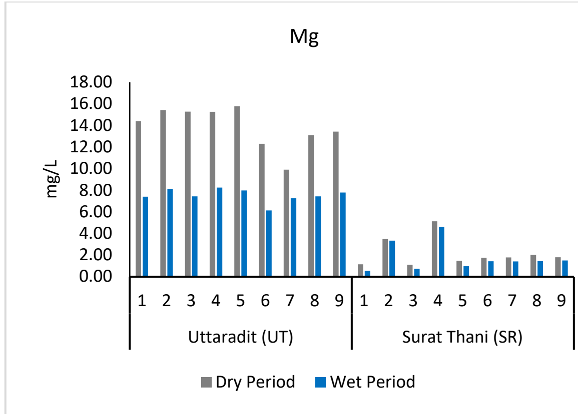

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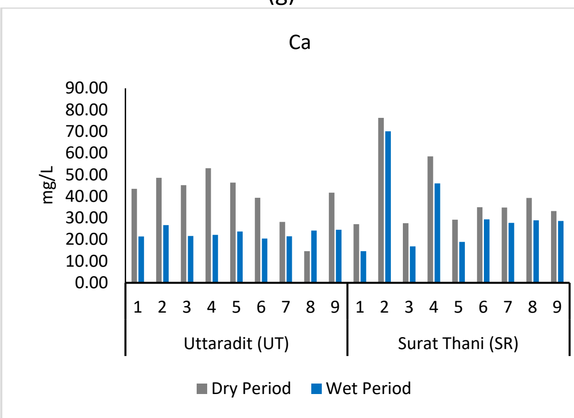

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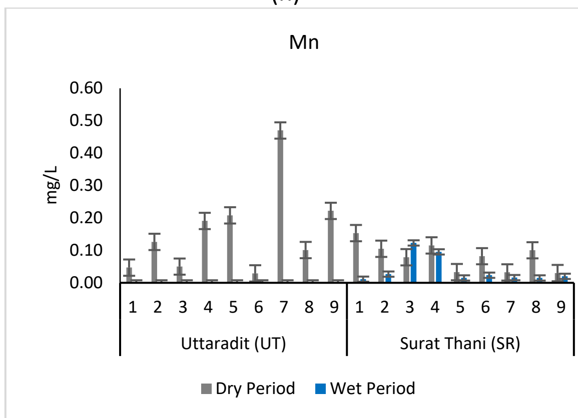

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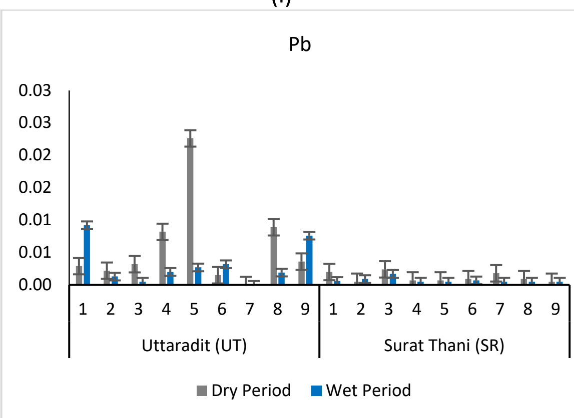

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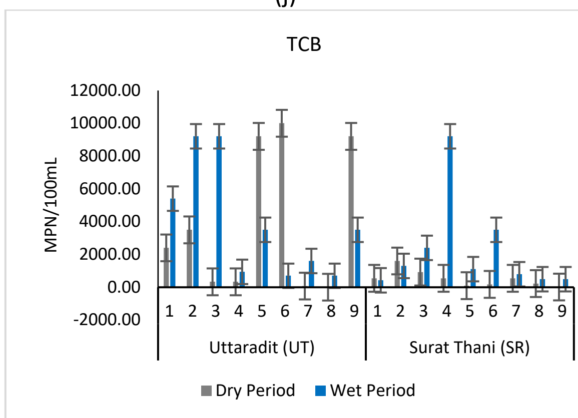
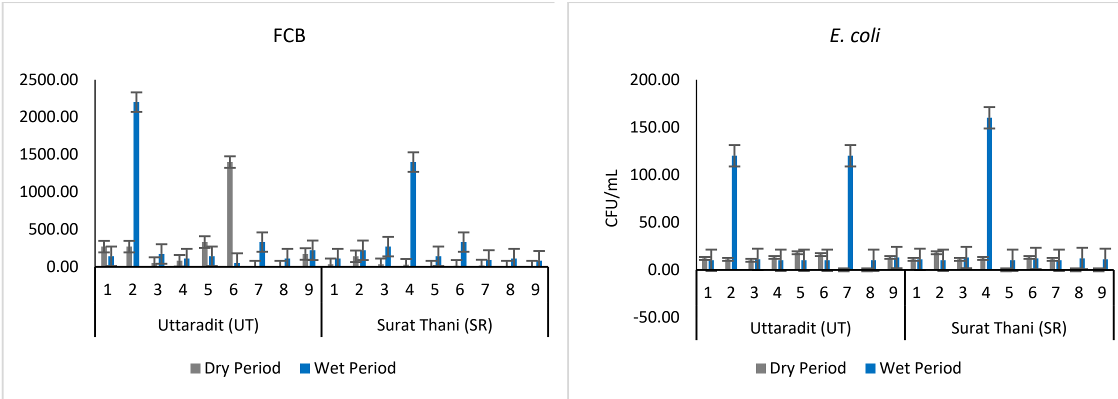


(l)

Continued Fig. 5: Water quality in Mae Phul–Mae Prong, Uttaradit and Klong Kram, Surat Thani during the wet and dry period. a) BOD; b) DO; c) TS; d) NO^{3}-; e) PO_4^{3-}; f) Fe; g) As; h) Ca; i) Mg; j) Mn; k) Pb; l) Total coliform bacteria; m) FCB; n) *E. coli*



(m) (n)

Continued Fig. 5: Water quality in Mae Phul–Mae Prong, Uttaradit and Klong Kram, Surat Thani during the wet and dry period. a) BOD; b) DO; c) TS; d) NO^{3-}; e) PO_4^{3-}; f) Fe; g) As; h) Ca; i) Mg; j) Mn; k) Pb; l) Total coliform bacteria; m) FCB; n) *E. coli*

Environment Board, 1994). The *t-test* results showed notable difference in water quality between wet and dry periods, particularly in the SR watershed for TS, phosphate, Ca, Mg, and Mn. In the UT watershed, significant differences were observed in pH, As, Ca, Mg, and Mn ($p < 0.05$). These variations can be attributed to the different parent material rocks, with limestone and granite predominating in the SR watershed and shale and mudstone in the UT watershed.

The stream water quality changed due to runoff and landslide processes such as typhoon-triggered landslides that delivered sediments to the downstream channels of the Shihmen Reservoir, Taiwan. Turbidity in the reservoir rapidly increased up to tenfold in the river catchment drainage every time, and the weight of landslide debris exceeded the total sediment discharge by fivefold (Lin *et al*., 2012). Moreover, land use and settlements were related to stream water quality (Cadwalader *et al*., 2011; Kiben *et al*., 2014; Palma *et al*., 2014). Although no clear empirical evidence for the dependency between slope stability and ion-loaded water was observed, the findings suggest that the flowing stream water might accelerate landslide movement (Preuth *et al*., 2010). In all these pollutants, freshwater could be detected for an extended period, where it could take more than 60 years for the entire deposit to erode naturally (Göransson *et al.,* 2018). External environmental factors, such as precipitation and location, as well as the internal environmental factors, such as the physical–chemical properties of the water, were closely related to the distribution of TCB, while FCB distribution was more associated with external inputs from seasonal runoff (Hong *et al.,* 2010). Although heavy-metal concentrations in the stream water in both the watersheds did not exceed the standard limits, except for the As concentration in the dry period at the station very close to the landslide scar, heavy-metal accumulation in the soil sediment was significantly different due to the specific characteristics of each heavy metal. Several previous studies have reported that heavy-metal concentrations were relatively high in the suspended compared to the bed sediment samples. Furthermore, for the wet season, the results revealed a positive correlation between As concentrations in surface water and fluvial sediments (Tapia *et al.,* 2022). Contamination sources in the bed and suspended sediment samples were natural processes and anthropogenic activities (Nazeer *et al.,* 2014). The processes that sequester trace elements can be grossly termed sorption, which to a large extent, determines the partitioning between the solid and solution phase. Factors such as soil pH, chemical speciation, fertility and soil amendments, redox potential, clay content, and soil structure, which affect trace element mobility and transport have been previously discussed (Carrillo-González *et al.*, 2006). The pH changes also influenced the suspended particles, particularly heavy-metal particles to be subsequently deposited, as they posed a toxicity risk to aquatic life and plankton (Usami *et al.,* 2019). Natural weathering of the As-rich solid phases affects As release, mainly from superficial soil horizons, with runoff contributing As to the surface water (Bossy *et al*., 2012). This process was further escalated owing to the biogeochemistry of the areas with elevated As concentration in the bedrock. In addition, higher heavy-metal concentration was observed during the wet season. (Azlini *et al.,* 2020). In water, As usually occurs in dissolved forms, with inorganic As being the most toxic (Barats *et al*., 2014). The dissolved As concentration in a river system depends on high hydrological variability and the drainage through different geological formations. The quantification of these As revealed that 79% was transported as particulate matter, with weathering contributing (58%–62%) more than geothermal spring discharge (38%–42%) to the total As content in river water (Zhou *et al.,* 2022). The highly heterogeneous distribution of As in draining through metamorphic rocks was influenced by the ore deposits containing arsenopyrite. Sedimentary rocks usually had low concentrations of As. Metamorphic rocks, including arsenopyrite, were located in proximity of the upland soil and were related to granite. As was dissolved and associated with natural organic matter, and the conditional distribution coefficients of As binding to the natural organic matter were relatively high (Neubauer *et al*., 2013). Although As and Pb were associated with similarly-sized particles, solid-phase As was more effectively mobilized and transported than Pb and Cd, which could pose a severe threat to the aquatic environment (Palma *et al*., 2014). This was attributed to the geological composition comprising granite and limestone, where arsenic discharge resulted from hydrogeochemical anomalies. These anomalies were promoted by chemical reactions between black shale and limestone, with pyrite oxidation leading to sulfuric acid formation. This accelerated rock weathering and discharge of heavy metals (Anda *et al.,* 2021; Sun *et al*., 2021; Del Gaudio *et al.,* 2024; Li *et al.,* 2024). Some species of heavy metals originate from various land use activities, e.g., the presence of chromium and nickel in soils were determined to be primarily due to iron ore mining. Agricultural activities were responsible for the observed concentration of Cu and Zn in soils (Samimi and Nouri, 2023). Relatively high concentrations of As were found in soils near gold mining activities (Luo *et al*., 2010). As found in the SR watershed may be related to granite rock and historical ore mining over the past 30 years, while that found in the UT watershed may be associated with shale rock and agricultural activities in upland areas. Analysis of water quality data from both watersheds in 2015 revealed no significant difference ($p > 0.05$), except for BOD concentrations. The UT stream water exhibited higher BOD concentrations than the SR watershed (1.45 ± 1.31 vs. 0.87 ± 0.46 mg/L), attributed to organic substance contamination due to mudflow processes during heavy rainfall and wastewater discharge from households near the stream water. The UT watershed had a larger number of households than the SR watershed. Comparing data between 2015 and 2022, BOD concentration in UT increased from 1.45 ± 1.31 mg/L in 2015 to 2.37 ± 0.12 mg/L in 2022 owing to population growth within the area. Conversely, the SR watershed had fewer households and minimal population growth. TCB levels increased from 4000.00 ± 379 to 4600.00 ± 325 most probable number per 100 mL (MPN/100mL) in UT and from 1300.00 ± 214 to 6500.00 ± 465 MPN/100mL in SR, while FCB increased from 370.00 ± 58 to 400.00 ± 67 MPN/100mL in UT and from 200.00 ± 34 to 600.00 ± 25 MPN/100mL in SR. Concentrations of Ca, Mg, Pb, and As in both watersheds were relatively similar in 2022 compared to 2015, with no significant differences observed. These findings align with those regarding the Ontake Lake, where water quality remained stable from 2015 to 2019 prior to a landslide (Usami *et al.,* 2019).

Elements in upland soil, sediment, and stream water

The correlations among the upland soil, sediment, and stream water demonstrate that the values of silt, As, and Mg concentrations in upland soil exhibited a significant correlation with the soil sediment and that the pH, Ca, Mg, and Mn concentrations in the soil sediment exhibited a correlation with stream water ($p < 0.05$). The results revealed that some heavy metals such as As were adsorbed by soil particles (Samimi *et al.,* 2023), particularly clay and silt particles, but some elements, including Ca and Mg, were dissolved and released from the soil sediment to the water body; which could increase the pH of the water (UT average = 7.84 ± 0.47 and SR average = 7.96 ± 0.34). The research findings align with heavy metal studies conducted in Iran and Indonesia. The sediment primarily contained Mn and Fe of natural origin, while Ca likely had an organic origin (Karbassi *et al*., 2015; Anda *et al.,* 2021). The average concentrations of As in the SR watershed were 121.0, 24.8, and 0.004 mg/L for the soil, sediment, and stream water, respectively, while those in the UT watershed were 0.025, 0.0015, and 0.003 mg/L, respectively. The stream water and sediment were affected by soil loss caused by landslide erosion (Rajasekar *et al.,* 2024). Key factors influencing heavy-metal pollution were related to slope, land use, and the predominant rock types, and the monsoon season (Reymond *et al.,* 2023). This implies that remedying existing landslides and reducing sediments in the watershed can considerably influence the potential of human water consumption and aquatic ecology (Tsai *et al*., 2012). Where the land use and land cover also played an important role in water quality changes (Dumago *et al.,* 2018).

CONCLUSION

The findings of this study reveal that the majority of particles in the upland soil and soil sediment samples comprise 80% sand. The soil in both the watersheds exhibits a slightly acidic nature and low organic matter content. Notably, the SR watershed displays considerably elevated concentrations of Ca and Mg, attributed to its limestone-based parent materials. Conversely, station No. 2 within the SR watershed, located in the landslide scar area, exhibited remarkably high levels of As owing to the geological composition comprising granite and limestone. Additionally, elevated levels of Pb, Mn, and Fe were observed in the UT watershed, resulting from the predominance of mudstone and shale as the parent materials in this region. Moreover, the biological quality of stream water revealed *E. coli* contamination at all sampling stations in both the watersheds, stemming from community activities such as wastewater discharge into the stream and sanitary tank leakage. Comparing water quality between wet and dry periods in the SR watershed revealed significant differences in SS, phosphate, Ca, Mg, and Mn. Conversely, in the UT watershed, differences were observed in pH and As, Ca, Mg, and Zn concentrations. These differences can be attributed to the geological composition of the respective watersheds. Although heavy-metal concentrations in the stream water of both the watersheds remained within standard limits, except for As concentration near the landslide scars during the dry period, heavy-metal accumulation in the soil sediment varied significantly owing to the different characteristics and environmental interactions of each metal. This study revealed the substantial impact of soil loss resulting from landslides on the quality of the soil sediment and stream water. The contamination of the stream water to the extent that it was unsuitable for direct human consumption emphasizes the critical role of considering arsenic contamination in landslide protection measures. The results of this study are valuable for the development and implementation of watershed plans aimed at protecting water resources and improving maintenance projects in prone areas to reduce costs and hazards. Moreover, the study highlights the importance of raising awareness among villagers and implementing effective mitigation strategies.

AUTHOR CONTRIBUTIONS

O. Phewnil conducted the literature review, experimental design, material preparation, data collection, analysis, and interpretation of data and prepared the manuscript text. O. Phewnil performed the study conception, experimental design, material preparation, data collection, analysis, manuscript text preparation, and manuscript editing. T. Pattamapitoon performed microbiological data collection and interpretation. N. Semvimol performed soil sediment data collection and interpreted the data. W. Wararam conducted the chemical analysis of the water samples. K. Duangmal performed upland soil sampling and analysis. A. Intaraksa performed land use and land cover mapping, K. Chunkao contributed to experimental design and manuscript editing. S. Hanthayung performed GIS mapping of the collection sites. P. Wichittrakarn performed data statistical analysis and interpretation. P. Maskulrath performed material preparation, data collection, and manuscript preparation.

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CONFLICT OF INTEREST

The authors declare that there is no conflicts of interest regarding the publication of this manuscript. In addition, the ethical issues, including plagiarism, informed consent, misconduct, data fabrication and/or falsification, double publication and/or submission, and redundancy have been completely observed by the authors.

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PUBLISHER'S NOTE

ABBREVIATIONS

%	Percent
°	Degree
°C	Degree Celsius
As	Arsenic
BOD	Biochemical oxygen demand
cm	Centimeter
Ca	Calcium
Ca^{2+}	Calcium ion
CFU/mL	Colony forming unit per milliliter
Cr	Chromium
Cu	Copper
DO	Dissolved oxygen
E	East
E. coli	Escherichia coli
EC	Electrical conductivity
FAAS	Flame atomic absorption spectrophotometer
FCB	Fecal coliform bacteria
Fe	Iron
kg	Kilograms
L	Liter
m	Meter
Mg	Magnesium
mg/kg	Milligram per kilogram
mg/L	Milligrams per liter
mL	milliliter
mm	Millimeter
Mn	Manganese
MPN/100mL	most probable number per 100 milliliters
N	North
N_2	Nitrogen
NH_3	Ammonia
NO_3^-	Nitrate
Pa	Pascal
Pb	Lead
pH	Potential of hydrogen
PO_4^{3-}	Phosphate
SR	Surat Thani province
TDS	Total dissolved solid
TS	Total solids
TN	Total Nitrogen
TCB	Total coliform bacteria
UT	Uttaradit Province

REFERENCES

Anda, M.; Purwanto, S.; Suryani, E.; Husnain; Muchtar, (2021). Pristine soil property and mineralogy as the strategic rehabilitation basis in post-earthquake-induced liquefaction, tsunami and landslide in Palu, Indonesia. Catena. 203: 105345 **(16 pages)**.

APHA, (2017). Standard Methods for Examination of Water and Wastewater, 23rd Ed. APHA-AWWA-WEF, Washington D.C.

Azlini, R.; Ismail, S.; Norkhadijah, S.; Suriyani, W.; Sarva Mangala, P.; Emilia, Z.A., (2020). The impact of seasonal change on river water quality and dissolved metals in mountainous agricultural areas and risk to human health. Environ. Forensics. 21(2): 195-211 **(7 pages).**

Barats, A.; Féraud, G.; Potot, C.; Philippini, V.; Travi, Y.; Durrieu, G.; Dubar, M.; SiCFUer, R., (2014). Naturally dissolved arsenic concentrations in the Alpine/Mediterranean Var River watershed (France). Sci. Total Environ., 473-474: 422–436 **(15 pages).**

Bathurst, J.C.; Bovolo, C.I.; Cisneros, F., (2010). Modelling the effect of forest cover on shallow landslides at the river basin scale. Ecol. Eng., 36(3): 317-327 **(11 pages).**

Bossy, A.; Grosbois, C.; Hendershot, W.; Beauchemin, S.; Crouzet, C.; Bril, H., (2012). Contributions of natural arsenic sources to surface waters on a high grade arsenic-geochemical anomaly (French Massif Central). Sci. Total Environ., 432: 257-268 **(12 pages).**

Carrillo-González, R.; Šimůnek, J.; Sauvé, S.; Adriano, D., (2006). Mechanisms and pathways of trace element mobility in soils. Adv. Agron., 91: 111-178 **(67 pages).**

Cadwalader, G.O.; Renshaw, C. E.; Jackson, B.P.; Magilligan, F. J.; Landis, J.D.; Bostick, B.C., (2011). Erosion and physical transport via overland flow of arsenic and lead bound to silt-sized particles. Geomorphology. 128: 85-91 **(7 pages).**

Chen, C.Y., (2009). Sedimentary impacts from landslides in the Tachia River Basin, Taiwan. Geomorphology. 105(3-4): 355-365 **(11 pages).**

Chiang, L.; Liao, C.; Lu C., (2021). Applicability of modified SWAT model (SWAT-TWN) on simulation of watershed sediment yields under different land use/cover scenarios in Taiwan. Environ Monit. Assess., 193: 520 **(23 pages)**.

Cheng, W.; Yang, G.; Yu, H.; Li, J.; Zhang, L., (2012). Impacts Wenchuan Earthquake induce landslide and soil physical properties and rice growth. Ecol. Indic., 15(1): 263–270 **(8 pages).**

Chuang, S.; Chen, H.; Lin, G.; Lin, C.; Chang, C., (2009). Increase in basin sediment yield from landslides in storms following major seismic disturbance. Eng. Geo., 103 (1-2): 59–65 **(7 pages).**

Damm, B.; Terhorst, B., (2010). A model of slope formation related to landslide activity in the Eastern Prealps, Austria. Geomorphology. 122 (3-4): 338–350 **(13 pages).**

Del Gaudio, E.; Steven Azzi, S.; Onorati, G.; Ducci, D., (2024). Changes in geochemical and isotopic contents in groundwater before seismic events in Ischia Island (Italy). Chemosphere. 349

Dumago, S.W.L.; Puno, G.R.; Ingotan, S.S., (2018). Water quality assessment in various land use and land cover of Muleta Watershed Bukidnon, Philippines. J. Biol. Environ. Sci., 12(3): 201-209 **(9 pages).**

Fan, S., (2017). Internal characteristics of loose solid source of debris Flow in Zhouqu. Sains Malaysiana. 46(11): 2179-2186 **(8 pages).**

Glade, T., (2003). Landslide occurrence as a response to land use change: a review of evidence from New Zealand. Catena. 51(3-4): 297–314 **(18 pages).**

Hong, H.; Qiu, J.; Liang, Y., (2010). Environmental Factors influencing the distribution of total and fecal coliform bacteria in six water storage reservoirs in the Pearl River Delta Region, China. J. Environ. Sci., 22 (5): 663–668 **(8 pages).**

Göransson, G.; Norrman, J.; Larson, M. (2018). Contaminated landslide run-out deposits in rivers – Method for estimating long-term ecological risks. Sci. Total Environ. 642: 553-566 **(14 pages).**

Jian, W.; Wang, Z.; Yin, K., (2009). Mechanism of the Anlesi landslide in the Three Gorges Reservoir, China. Eng. Geo., 108 (1): 86–95 **(10 pages).**

Jotisankasa, A.; Mairaing, W.; Tansamrit, S., (2020). Infiltration and stability of soil slope with vetiver grass subjected to rainfall from numerical modeling. In Russell, A.R. (Eds.), Unsaturated soils: Research and Applications. CRC Press., 1241-1247 **(7 pages).**

Karbassi, A.R.; Pazoki, M., (2015). Environmental Qualitative assessment of rivers sediments. Global J. Environ. Sci. Manage. 1 (2): 109-116 **(8 pages).**

Kiben, J.; Nhapi, I.; Gumindoga, W., (2014). Assessing the relationship between water quality parameters and changes in landuse patterns in the upper Manyame River, Zimbabwe. Phy. Chem. Earth., 67–69: 153–163 **(11 pages).**

Kondratyeva, L.M.; Makhinov, A.N.; Andreeva, D.V.; Bashkurova, A.S. (2020). Changes in water quality in the Bureiskoe reservoir caused by a large landslide. Water Resour. 47: 257–268 **(12 pages).**

Korkanc, S.Y.; Ozyuvaci, N.; Hizal, A., (2008). Impacts of land use concession on soil properties and soil erodibility. J. Environ. Biol., 29 (3): 363–370 **(8 pages).**

La Licata, M.; Bosino A.; Bettoni, M.; Maerker, M., (2023). Assessing landscape features and geomorphic processes influencing sediment dynamics in a geomorphologically highly active Mediterranean agroecosystem: The upper Val d'Arda case study (Northern Apennines, Italy). Geomorphology. 433: 108724 **(22 pages).**

Lin, G.W.; Chen, H., (2012). The relationship of rainfall energy with landslides and sediment delivery. Eng. Geo., 125: 108–118 **(11 pages).**

Li, D.; Zhang, H.; Chang, F.; Duan, L.; Zhang, Y., (2024). Environmental arsenic (As) and its potential relationship with endemic disease in southwestern China. J. Environ. Sci. 139: 46-59 **(14 pages)**.

Luo, W.; Lu, Y.; Zhang, Y.; Fu, W.; Wang, B.; Jiao, W.; Wang, G.; Tong, X.; Giesy, J.P., (2010). Watershed-scale assessment of arsenic and metal contamination in the surface soils surrounding Miyun Reservoir, Beijing, China. J. Environ. Manage., 91 (12): 2599–2607 **(8 pages).**

Mark, O.; Timothy, O.; Christopher, A.; Emmanuel, N., (2024). Spatio-temporal variations and potential health risks of heavy metals in water from river Manafwa, Uganda. Lett. Appl. Nano. Bio. Sc., 13(1): 1-27 **(27 pages)**.

Mouri, G.; Shiiba, M.; Hori, T.; Oki, T., (2011). Modeling shallow landslides and river bed variation associated with extreme rainfall-runoff events in a Granitoid mountainous forested catchment in Japan. Geomorphology. 125 (2): 282–292 **(11 pages).**

National Environment Board, (1994). Standard of Surface Water Quality of Thailand: No. 8, B.E.2537 (1994), issued under the Enhancement and Conservation of National Environmental Quality Act B.E.2535 (1992) dated January 20, B.E.2537 (1994). The Royal Government Gazette, Thailand.

Nazeer, S.; Hashmi, M.Z.; Malik, R.N., (2014). Heavy metals distribution, risk assessment and water quality characterization by water quality index of the river Soan, Pakistan. Ecol. Indic., 43: 262–270 **(9 pages).**

Neubauer, E.; Kammer, F.V.D.; Knorr, K.H.; Peiffer, S.; Reichert, M.; Hofmann, T., (2013). Colloid-associated export of arsenic in stream water during storm flow events. Chem. Geol., 352: 81–91 **(11 pages).**

Okalebo, J.R.; Gathua, K.W.; Woomer, P.L., (2002). Laboratory Methods of Soil and Plant Analysis: A working Manual, 2nd Ed. TSBF-CIAT and SACRED Africa, Kenya.

Opiso, E.M.; Puno, G.R.; Alburo, J.LP; Detalla, A.L., (2016). Landslide susceptibility mapping using GIS and FR method along the Cagayan de Oro-Bukidnon-Davao City route corridor, Philippines. KSCE J Civ Eng. 20: 2506–2512 **(13 pages).**

Palma, P.; Ledo, L.; Soares, S.; Borlosa, I.R.; Aluarenga, P., (2014). Spatial and temperal variability of the water and sediments quality in the aquifer reservoir (Guadiana Basin, Southern Portugal). Sci. Total Environ., 470–471: 780–790 **(11 pages).**

Paolini, L.; Villalba, R.; Ricardo, Grau, H.R., (2005). Precipitation variability and landslide occurrence in a subtropical mountain ecosystem of NW Argentina. Dendrochronologia., 22 (3): 175–180 **(6 pages).**

Payus, C.; Anuar, S.S.I.; Musta, B.; Bidin, K.; Tongkul, F.; Moh, P.Y., (2023). Water quality for river basins after post-earthquake event. Environ. Ecol. Res., 11(1): 218-223 **(6 pages).**

Priyono, P.; Handoko, S.; Rahayu, R.; Sawitri, S.; Murwaningtyas, F.S.D., (2022). Determining mitigation on landslide prone level in watershed area: analysis of study on Samin Hulu Karanganyar District of Indonesia and its economic consequences. Econ. Annals-XXI. 199(9-10): 40-47 **(8 pages).**

Preuth, T.; Glade, T.; Demoulin, A., (2010). Stability analysis of a human-influenced landslide in eastern Belgium. Geomorphology. 120 (1-2): 38–47 **(10 pages).**

Rajasekar, A.; Murava, R.T.; Norgbey, E.; Zhu, X., (2024). Spatial distribution, risk index, and correlation of heavy metals in the Chuhe River (Yangtze tributary): preliminary research analysis of surface water and sediment contamination. Appl. Sci., 14: 904 **(20 pages).**

Reymond, D.J.; Sudalaimuthu, K., (2023). Geospatial visualization and seasonal variation of heavy metals in river sediments. Global J. Environ. Sci. Manage. 9(2): 309-322 **(14 pages).**

Rijsdijk, A.; Sampurno, Bruijnzeel L.A.; Sutoto, C.K., (2007). Production of runoff and sediment yield from rural road, trail, and attlements in the upper kanto catchment, East Java, Indonesia. Geomorphology. 87(1-2): 28–37 **(10 pages).**

Samimi, M.; Nouri, J., (2023). Optimized Zinc Uptake from the Aquatic Environment Using Biomass Derived from Lantana Camara L. Stem, Pollution, 9(4): 1925-1934 **(10 pages).**

Samimi, M.; Zakeri, M.; Alobaid, F.; Aghel, B., (2023). A Brief Review of Recent Results in Arsenic Adsorption Process from Aquatic Environments by Metal-Organic Frameworks: Classification Based on Kinetics, Isotherms and Thermodynamics Behaviors. nanomaterials, 13(1): 60 **(12 pages).**

Sanghapi, H.K.; Jain, J.; Bol'shakov, A.; Lopano, C.; McIntyre, D.; Russo, R., (2016). Determination of elemental composition of shale rocks by laser induced breakdown spectroscopy (LIBS). Spectrochimica Acta Part B: At. Spectrosc., 122: 9-14 **(6 pages).**

Sewell, R.J.; Parry, S.; Millis, S.W.; Wang, N.; Rieser, U.; Dewitt, R., (2015). Dating of debris flow fan complexes from Lantau Island, Hong Kong, China: the potential relationship between landslide activity and climate change. Geomorphology. 248: 205–227 **(22 pages).**

Shanmugam, G., (2015). The landslide problem. J. Paleogeogr., 4(2): 109-166 **(58 pages).**

Sun, C.; Ling, S.; Wu, X.; Li, X.; Chen, J.; Jiang, W., (2021). Oxidation of black shale and its deterioration mechanism in the slip zone of the Xujiaping landslide in Sichuan Province, Southwestern China. Catena. 200: 105139 **(14 pages).**

Tapia, J.; Mukherjee, A.; Rodríguez, M.; Murray, J.; Bhattacharya, P., (2022). Role of tectonics and climate on elevated arsenic in fluvial systems: Insights from surface water and sediments along regional transects of Chile. Environ. Pollut., 314.

Trincsi, K.; Pham, T.; Turner, S., (2014). Mapping mountain diversity: ethnic minorities and land use land cover change in Vietnam's borderlands. Land Use Policy., 41: 484–497 **(14 pages).**

Tsai, Z.; Yan, G.J.; Lee, H.; Chin, Y., (2012). Use of a total station to monitor pool-failure sediment yields in landslide with of the Shihmen resource watershed, Taiwan. Geomorphology. 139–140: 438–451 **(14 pages).**

Turner, T. R.; Duke S. D.; Fransen, B. R.; Reiter, M. L.; Kroll, A. J.; Ward, J. W.; Bach, J. L.; Justice, T. E; Bilby, R. E., (2010). Landslide densities associated with rainfall, stand age, and topography on forested landscapes, southwestern Washington, USA. For. Ecol. Manage., 259: 2233-2247 **(15 pages).**

Uchida, E.; Osada, K.; Nakao, K., (2017). Major and trace element chemical compositional signatures of some granitic rocks related to metal mineralization in Japan. Open J. Geol., 7: 559-576 **(18 pages).**

USEPA, (1996). Method 3050B: Acid Digestion of Sediments, Sludges, and Soils. 2nd Ed. Washington, DC. **(12 pages).**

Usami, A.; Matsumoto, Y.; Yagi, A.; Iwatsuki, E. (2019). Role of difference among rivers affected by volcanic activities of MT. Ontake for water quality of the Nigorigawa River. Geotec., Const. Mat. Environ., 58: 76-81 **(6 pages).**

Yoshihara,N.; Matsumoto, S.; Umezawa, R.; Machida, I., (2022). Catchment-scale impacts of shallow landslides on stream water chemistry. Sci. Total Environ., 825: 153970 **(10 pages)**.

Zhen, N.; Fu, B.; Lü, Y.; Zheng, Z., (2014). Changes of livelihood due to land use shifts: A case study of Yanchang County in the Loess Plateau of China. Land Use Policy. 40: 28–35 **(8 pages).**

Zhou, Li,; Xu, Z.; Zhou, J.; Fan, P., (2022). Natural arsenic source, migration, and flux in a catchment on the Southern Tibetan Plateau. Sci. Total Environ., 838(2).

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