ECOLOGICAL RISK ASSESSMENT OF HEAVY METALS IN SOILS FROM AN INFORMAL E-WASTE DISMANTLING SITE, BURIRAM PROVINCE, THAILAND

Miss Nisakorn Amphalop

A Thesis Submitted in Partial Fulfillment of the Requirements
for the Degree of Master of Science in Hazardous Substance and
Environmental Management
Inter-Department of Environmental Management
GRADUATE SCHOOL
Chulalongkorn University
Academic Year 2019

การประเมินความเสี่ยงต่อระบบนิเวศของการปนเปื้อนโลหะหนักในดินบริเวณชุมชนรื้อแยกขยะอิเล็กทรอนิกส์ จังหวัดบุรีรัมย์ ประเทศไทย

น.ส.นิสากร อัมพะลพ

วิทยานิพนธ์นี้เป็นส่วนหนึ่งของการศึกษาตามหลักสูตรปริญญาวิทยาศาสตรมหาบัณฑิต
สาขาวิชาการจัดการสารอันตรายและสิ่งแวดล้อม สหสาขาวิชาการจัดการสิ่งแวดล้อม
บัณฑิตวิทยาลัย จุฬาลงกรณ์มหาวิทยาลัย
ปีการศึกษา 2562

Thesis Title	ECOLOGICAL RISK ASSESSMENT OF HEAVY METALS IN SOILS FROM AN INFORMAL E-WASTE DISMANTLING SITE, BURIRAM PROVINCE, THAILAND
By	Miss Nisakorn Amphalop
Field of Study	Hazardous Substance and Environmental Management
Thesis Advisor	Assistant Professor TASSANEE CHETWITTAYACHAN, Ph.D.
Thesis Co Advisor	Mongkolchai Assawadithalerd, Ph.D.

Accepted by the GRADUATE SCHOOL, Chulalongkorn University in Partial Fulfillment of the Requirement for the Master of Science

.... Dean of the GRADUATE SCHOOL
(Assistant Professor THUMNOON NHUJAK, Ph.D.)

THESIS COMMITTEE

.... Chairman
(Associate Professor EKAWAN LUEPROMCHAI, Ph.D.)
.... Thesis Advisor
(Assistant Professor TASSANEE CHETWITTAYACHAN, Ph.D.)
.... Thesis Co-Advisor
(Mongkolchai Assawadithalerd, Ph.D.)
.... Examiner
(Assistant Professor PENRADEE CHANPIWAT, Ph.D.)
.... External Examiner
(Associate Professor Tunlawit Satapanajaru, Ph.D.)

นิสากร อัมพะลพ : การประเมินความเสี่ยงต่อระบบนิเวศของการปนเปื้อนโลหะหนักในดินบริเวณชุมชนรื้อแยกขยะอิเล็กทรอนิกส์ จังหวัดบุรีรัมย์ ประเทศไทย. (ECOLOGICAL RISK ASSESSMENT OF HEAVY METALS IN SOILS FROM AN INFORMAL E-WASTE DISMANTLING SITE, BURIRAM PROVINCE, THAILAND) อ.ที่ปรึกษาหลัก : ผศ. ดร.ทรรศนีย์ พฤกษาสิทธิ์, อ.ที่ปรึกษาร่วม : ดร.มงคลชัย อัศวดิษฐเลิศ

งานวิจัยนี้มีวัตถุประสงค์เพื่อศึกษาการปนเปื้อน รูปฟอร์มของโลหะหนักและการประเมินความเสี่ยงต่อระบบนิเวศของการปนเปื้อนโลหะหนักในดินบริเวณชุมชนรื้อแยกขยะอิเล็กทรอนิกส์ ตำบลแดงใหม่และตำบลบ้านเป้า จังหวัดบุรีรัมย์ โดยดินตัวอย่างถูกเก็บมาจากบริเวณที่มีกิจกรรมการคัดแยกขยะอิเล็กทรอนิกส์ซึ่งประกอบด้วยบริเวณบ้านที่มีกิจกรรมการคัดแยกขยะอิเล็กทรอนิกส์และบริเวณจุดเผาขยะอิเล็กทรอนิกส์ และบริเวณพื้นที่ใกล้เคียงซึ่งประกอบด้วยดินนาข้าวที่ตั้งอยู่รอบบริเวณจุดเผา ดินบริเวณบ้านที่ไม่มีกิจกรรมการคัดแยกและดินบริเวณบ่อน้ำใต้ดิน โดยดินที่กำหนดให้เป็นจุดอ้างอิงอยู่ไกลจากบริเวณจุดคัดแยกขยะอิเล็กทรอนิกส์ออกไป 5 กิโลเมตร จากนั้นย่อยตัวอย่างดินด้วยกรดไนตริกผสมกับกรดไฮโดรคลอริกด้วยเครื่องไมโครเวฟและสกัดรูปฟอร์มของโลหะหนักโดยใช้วิธีการสกัดลำดับขั้นตอนแบบบีซีอาร์ดัดแปลง จากนั้นตรวจปริมาณของโลหะหนักทั้ง 8 ชนิด (สารหนู แคดเมียม โครเมียม ทองแดง แมงกานีส นิกเกิล ตะกั่ว และสังกะสี) ด้วยเครื่อง inductively coupled plasma-optical emission spectroscopy (ICP-OES) ผลการวิจัยพบว่าปริมาณแคดเมียม ทองแดง ตะกั่ว และสังกะสีในดินบริเวณจุดเผาสูงกว่าในดินบริเวณจุดอ้างอิงและจุดอื่นๆ อย่างมีนัยสำคัญที่ระดับความเชื่อมั่น 95% แสดงให้เห็นว่ากิจกรรมการเผาขยะอิเล็กทรอนิกส์ก่อให้เกิดการปนเปื้อนโลหะในดินอย่างมีนัยสำคัญ โดยปริมาณสารหนู ทองแดง ตะกั่วและสังกะสี เกินค่ามาตรฐานเนเธอร์แลนด์ (intervention value) ที่บ่งบอกถึงการปนเปื้อนระดับที่สูงและควรได้รับการพิจารณาเพื่อบำบัด ปริมาณโลหะหนักทุกชนิดในดินนาข้าวมีความเข้มข้นรองลงมาจากที่พบในดินบริเวณจุดเผาขยะอิเล็กทรอนิกส์ แสดงให้เห็นว่ากิจกรรมการเผาขยะอิเล็กทรอนิกส์ส่งผลต่อการปนเปื้อนโลหะหนักในดินของบริเวณโดยรอบ ผลการศึกษารูปฟอร์มของโลหะหนักในดินพบว่า การกระจายตัวของสารหนูและนิกเกิลส่วนใหญ่อยู่ในรูปฟอร์มที่ถูกออกซิไดซ์ได้ (oxidizable fraction) แคดเมียม ทองแดงและตะกั่วส่วนใหญ่อยู่ในรูปฟอร์มที่ถูกรีดิวซ์ได้ง่าย (reducible fraction) สังกะสีและแมงกานีสส่วนใหญ่จะอยู่ในรูปที่แลกเปลี่ยนได้ (exchangeable/acid soluble fraction) ผลจากการประเมินความเสี่ยงต่อระบบนิเวศด้วยวิธี Potential Ecological Risk Index (PERI) และ Risk Assessment Code (RAC) ที่พิจารณาความเข้มข้น ความเป็นพิษและความสามารถที่จะเคลื่อนที่ไปสู่สิ่งมีชีวิตของโลหะหนัก พบว่าแคดเมียม ทองแดง ตะกั่ว นิกเกิล และสังกะสีก่อให้เกิดค่าความเสี่ยงสูงในระบบนิเวศของบริเวณจุดเผาขยะอิเล็กทรอนิกส์ ขณะที่ทองแดงก่อให้เกิดค่าความเสี่ยงสูงในระบบนิเวศบริเวณนาข้าวที่อยู่ติดบริเวณจุดเผาขยะอิเล็กทรอนิกส์

สาขาวิชา	การจัดการสารอันตรายและสิ่งแวดล้อม	ลายมือชื่อนิสิต ....
ปีการศึกษา	2562	ลายมือชื่อ อ.ที่ปรึกษาหลัก ....
		ลายมือชื่อ อ.ที่ปรึกษาร่วม ....

6187527620 : MAJOR HAZARDOUS SUBSTANCE AND ENVIRONMENTAL MANAGEMENT

KEYWORD: E-waste, Heavy metals, Ecological risk assessment, Heavy metal fractionation

Nisakorn Amphalop : ECOLOGICAL RISK ASSESSMENT OF HEAVY METALS IN SOILS FROM AN INFORMAL E-WASTE DISMANTLING SITE, BURIRAM PROVINCE, THAILAND. Advisor: Asst. Prof. TASSANEE CHETWITTAYACHAN, Ph.D. Co-advisor: Mongkolchai Assawadithalerd, Ph.D.

The research aimed to investigate the contamination and fractionations, and evaluate the ecological risk posed by heavy metals in the e-waste site in Daeng Yai and Ban Pao e-waste dismantling sites, Buriram Province, Thailand. The soil samples were taken from e-waste houses (EW), and e-waste open-burning sites (OB) as representatives of the e-waste sites, while soils from paddy fields (PF), non-e-waste houses, and groundwater wells represented the surrounding area, and the reference site was located 5 km away from the e-waste dismantling site. The soil samples were digested by aqua regia by the microwave digester for total heavy metal determination, and the metals in soils were sequentially extracted following the modified BCR sequential extraction procedure. The contents of As, Cd, Cr, Cu, Mn, Ni, Pb, and Zn were analyzed by Inductively Coupled Plasma-Optical Emission Spectroscopy (ICP-OES). The concentrations of Cd, Cu, Pb, and Zn in OB were significantly higher ($p<0.05$) than the control site, suggesting the significant impacts of the e-waste recycling activities at OB. As, Cu, Pb, and Zn in the OB were exceeded the soil intervention value, the representative of the severe case of soil contamination, which indicated that the OB was seriously contaminated and remediation would be needed. Metal contents in soils collected from PF were secondly high compared to those from OB, suggesting that PF closely located to the OB could be contaminated by the metals emitted from open-burning activities. Applying the modified BCR procedure, As and Ni mostly presented in an organic bound fraction. Cd, Cu, and Pb were dominantly associated with Fe-Mn oxides fractions. Zn and Mn were mainly presented in an exchangeable and acid soluble fraction. The results indicated that Zn and Mn had the highest potential bioavailability and mobility in the site. For the ecological risk assessment based on the heavy metal contents, toxicity, and mobility, e-waste activities in the OB and EW have potentially posed ecological risk caused by Cd, Cu, Pb, Ni, and Zn. In addition, Cu posed potential ecological risk to the PFs, the surrounding area of the OB.

Field of Study:	Hazardous Substance and Environmental Management	Student's Signature ....
Academic Year:	2019	Advisor's Signature ....
		Co-advisor's Signature ....

ACKNOWLEDGEMENTS

Foremost, I would like to express my gratitude to my thesis advisor and co-advisor, Assistant Professor Dr. Tassanee Prueksasit, and Dr. Mongkolchai Assawadithalerd for continuing support and giving useful advises, encouragement, patience, and immense knowledge.

Moreover, I would like to thank my thesis committee: Associate Professor Dr. Ekawan Luepromchai, Associate Professor Dr. Tunlawit Satapanajaru, and Assistant Professor Dr. Penradee Chanpiwat for giving insightful comments and suggestion to completion the thesis.

I would like to acknowledge Associate Professor Dr. Apichat Imyim, and Dr. Kasitnun Chayavanich, Department of Chemistry, Chulalongkorn University, for helping with the heavy metal analysis by ICP-OES. My sincere thanks to laboratory staffs of environmental science department, and center of excellence on hazardous substance management, Chulalongkorn University, for the equipment of laboratory, and helpful support.

My grateful thanks are also extended to local public health officers and the villagers at Ban Pao, and Daeng Yai for kindly supporting during the fields sampling. Also, I am thankful for the kind support from my friends.

Last but not least, I would like to thank you my family for the endless support in every way, encouragement throughout the study. Thanks to everyone who helps but have not been mentioned in this acknowledgement.

Nisakorn Amphalop

TABLE OF CONTENTS

Page

LIST OF TABLES

Page

LIST OF FIGURES

Page

Chapter 1

Introduction

1.1 Rationale background and problem addressed

E-waste is a term used to define “items of all types of electrical and electronic equipment and its parts that have been discarded by the owners as waste without the intention of re-use” (UNU, 2014). E-waste generation continues to be a global concern since the rapidly growing of technological advancement in electronic equipment has resulted in more people replacing their old equipment with new ones. Furthermore, the decreasing costs of electronic equipment have led more purchases (Leung, 2019). In Thailand, e-waste was growing rapidly with increasing rate at least 7-8% between 2012-2015 (Withayaanumas, 2017). By 2020, e-waste generation of computers has been forecasted to reach approximately 7.5 million units, while industrial e-waste has been estimated to be around 11,000 tons per year (Manomaivibool & Vassanadumrongdee, 2011). In addition, the amount of 13.42 million mobile phones, and 3.65 million portable audio players have been expected to be found in 2021 (Withayaanumas, 2017).

Few of the e-waste is collected and recycled properly, but a large proportion of them are disposed and transported to family-run e-waste recycling/dismantling workshops where sections of household area are used to store and dismantle e-waste in order to obtain precious metals and valuable materials for sale. The techniques used to dismantle and recycle e-waste in the workshops are carried out by primitive dismantling methods. These include physical dismantling to disassembly e-waste like smash, cut, break, open-burning of wire cables to recover copper, split air-conditioner compressors to extract copper and ferrous, and open-burning of abandoned components. After sorting, the precious materials are sold, while the unwanted and residue waste are discarded in the open dump site of municipal solid waste (Thongkaow et al., 2017; Vassanadumrongdee, 2014).

Electronic equipment contains valuable materials such as Pt, Cu, Au, Fe, Al, plastic, and glass which can be separated and sold as secondary raw materials. Meanwhile, various toxic metals like Cd, Cr, Cu, Pb, and Hg are also contained in the equipment. The uncontrolled recycling and informal dismantling activities have caused the release of the metal contaminating in soils at e-waste dismantling sites, e-waste burning sites and also in surrounding e-waste sites like pond areas, paddy fields, and vegetable gardens through atmospheric deposition, and runoff (Jun-hui & Hang, 2009; Luo et al., 2011; Quan et al., 2015; Tang et al., 2010). For example, according to Luo et al. (2011), the soils in former e-waste incineration site in Guangdong Province, China were highly contaminated by Cd (17.1 mg/kg), Cu (11,140 mg/kg), Pb (4,500 mg/kg), and Zn (3,690 mg/kg), while the soils of nearby paddy fields and vegetable gardens also had relatively high contamination of Cd (1 and 0.9 mg/kg, respectively) and Cu (155 and 324 mg/kg, respectively), which exceeded the Chinese standard for agricultural soil of 0.3 mg/kg Cd and 50 mg/kg of Cu.

In the past several years, there have been dismantling activities of e-waste in Daeng Yai subdistrict and Ban Pao subdistrict, Buriram Province, Thailand. The e-waste dismantling activities are normally operated at household areas and dump sites. The activities carried out at the household area consist of smashing and separating of electric motors, printed circuit boards contained in washing machines, refrigerators and air-conditioners in order to get copper and steel. The burning of wires and breaking of cathode ray tube (CRT) screens aiming to recover copper, steel, and aluminum are implemented at the dump sites, where the glass, insulation foams, and other unwanted materials are dumped on land. The previous study investigated the heavy metal contamination at the e-waste dismantling house, and detected elevated levels of As (0.01-0.38 mg/kg), Cu (1.18-380.40 mg/kg) and Pb (0.11-15.28 mg/kg) in the soils (Amphalop et al., 2020; Thongkaow et al., 2017). As e-waste houses are sparsely located in the villages, and dump sites are located in the field adjacent to land used for agricultural purposes, heavy metals from informal open burning and dismantling of e-waste could penetrate the soils where rice and crops are grown. As a result, information on heavy metal contaminations is necessary to be investigated in the vicinity of the e-waste site. However, elevated concentrations of heavy metals cannot clearly assess the potential availability and consequences of heavy metals on plants, animals and ecosystems in the area (Isimekhai et al., 2017).

Sequential extraction procedure is a tool to evaluate the mobility and potential availability of metals in soils to plant. Based on the BCR (Community Bureau of Reference) sequential extraction procedure, the partitioning of the heavy metals by their associations with phases or fractions characterizes the heavy metals into acid-soluble phase, reducible phase, oxidizable phase, and residual phase (Rauret et al., 1999). An acid-soluble phase is made up of exchangeable metal and bound to carbonates. It is easily dissolved and taken up by plants and organisms. The reducible phase consists of metals bound to Fe and Mn oxides that can be released depending on the soil condition. The oxidizable phase is metals bound to organic matter which can be mobilized with time and decomposition. Reducible and Oxidizable are potentially mobile and bioavailable. The residual phase is tightly bound within the soil structure, so it is not usually available to biota. With the potential bioavailability, toxicity and mobility of metals in soils, these four fractions, in turn, allow the assessment of the risk posed by them (Perin et al., 1985; Sarkar et al., 2014).

The total heavy metal contamination and the concentrations of metals associated with the specific fractions have been noted to be affected by change in season, depth of soil and soil properties (Amphalop et al., 2020; Olafisoye et al., 2013). It has been reported that change in environmental conditions, such as pH, redox potential, organic matter, and cation exchange could affect adsorption and precipitation that eventually cause the metal released from solid to liquid phase (Rieuwerts et al., 1998). For this reason, the factors affecting heavy metals in soils are necessary to be investigated to give a better understanding of the soil conditions influencing the bioavailable of heavy metals in the specific area.

Up to now, for Daeng Yai and Ban Pao e-waste dismantling sites, there has been no study focusing on the heavy metal contamination in the surrounding areas of the e-waste sites, heavy metals mobility and bio-availability, and factor affecting the heavy metal concentration in such areas. Therefore, the ecological risk from heavy metals is the major concerns of this study, and the aims of this study are included; (1) to investigate the total contamination of heavy metals in soils at the e-waste site and surrounding areas. (2) to investigate the fractions of the heavy metals in the e-waste site and surrounding environment using the modified BCR sequential extraction approach. (3) to characterize the soil properties that govern the phase associations of heavy metals in certain soils from the e-waste site and surrounding areas, and (4) to evaluate the potential risk posed to the e-waste site and surrounding environment as a result of the e-waste dismantling activities.

1.2 Objectives

The main objective of the study is to evaluate the potential risk posed by heavy metals in the e-waste site and surrounding areas as a result of the e-waste dismantling; there are three sub-objectives in the study as follows:

(1) To investigate the total contamination of heavy metals in soils at the e-waste site and surrounding areas.

(2) To investigate the fractions of the heavy metals in the e-waste site and surrounding areas using a sequential extraction approach.

(3) To evaluate the ecological risk posed by heavy metals in the e-waste site and surrounding areas.

1.3 Hypotheses

(1) Soils collected from the e-waste site and surrounding areas will be found to have a higher concentration of heavy metals than those collected from the control area where an e-waste dismantling activity has not been operated.

(2) The percentage of each fraction of heavy metal will be different among soil sampling sites as a result of the differences in soil properties among the sampling sites.

(3) The ecological risk at the e-waste site will indicate a higher risk level compared to the surrounding and the control area.

1.4 Scope of study

The soil samples were collected from Daeng Yai and Ban Pao e-waste dismantling sites, Buriram Province, Thailand. The soil samples were taken at the depth of 0-15 cm and 15-30 cm from the e-waste sites (e-waste dismantling houses and e-waste burning sites) and surrounding areas (paddy fields, non-e-waste dismantling houses, and groundwater well area). It was conducted during two seasons, which were the dry (April 2019) and wet (September 2019) season. The total concentration of As, Cd, Cr, Cu, Mn, Ni, Pb, and Zn was determined using inductively coupled plasma-optical emission spectroscopy. In addition, the fractionation of the heavy metals was investigated using modified BCR sequential extraction. After that, the ecological risk posed by the heavy metals was assessed using the potential ecological risk assessment approach and the risk assessment code approach. The overall research framework for this study is shown in Figure 1.

1.5 Expected outcomes

(1) To obtain the heavy metal contaminations in the study area as a result of the informal e-waste dismantling activities

(2) To understand the availability and mobility of the heavy metals in the soil reflecting the dissemination of the heavy metals from the e-waste sites to surrounding areas.

(3) To obtain the potential risk levels posed to the environment in the e-waste site showing the consequences of the heavy metals on the ecosystem of the study site.

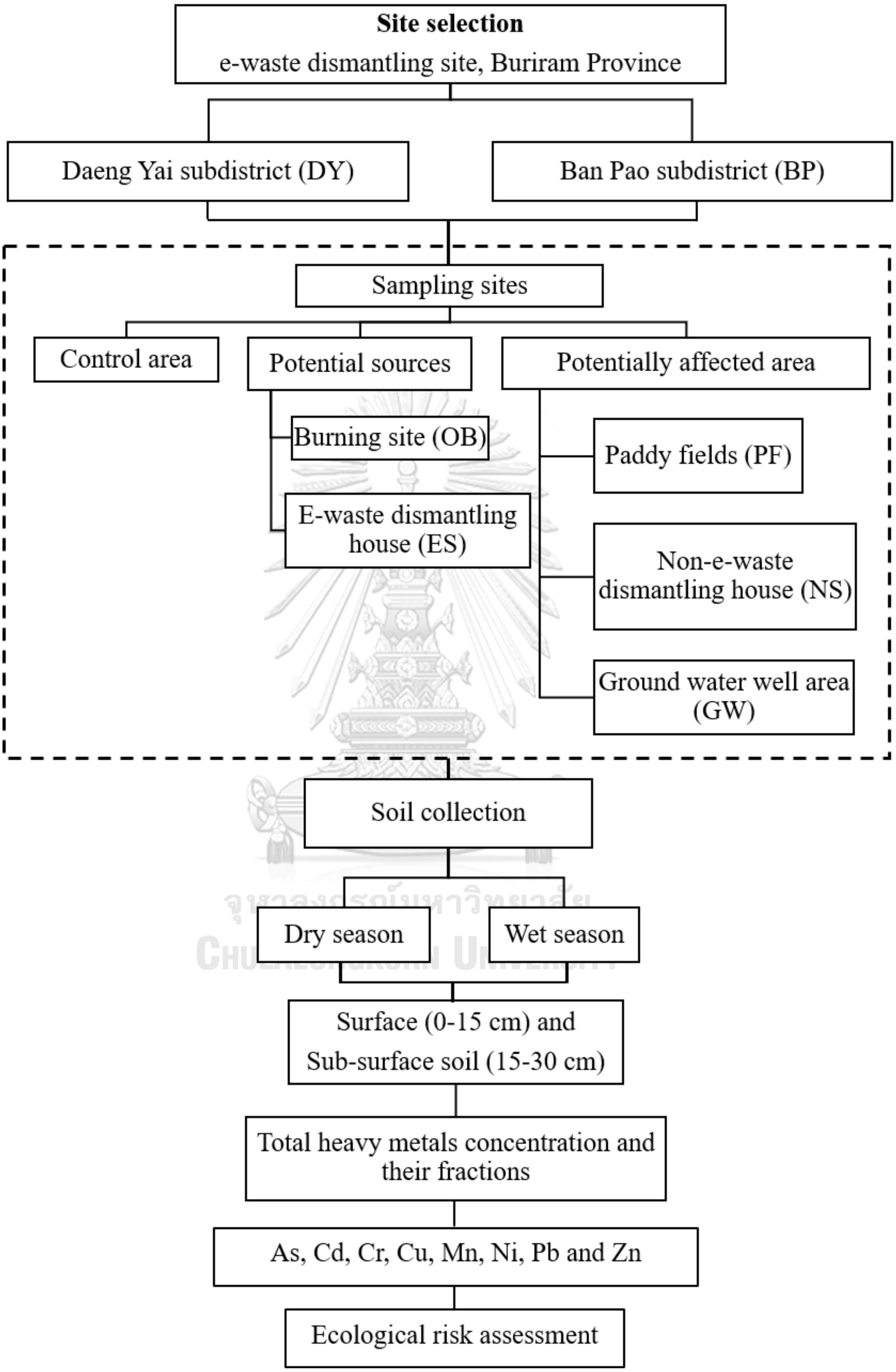


Figure 1 Research conceptual framework for this study

Chapter 2

Literature review

2.1 Definition of e-wastes

E-waste, also referred to waste electrical and electronic equipment (WEEE), is a term used to define "items of all types of electrical and electronic equipment and its parts that have been discarded by the owners as waste without the intention of re-use". E-waste also includes small equipment like personal computers and mobile phones and large equipment such as washing machines, dryers, and air conditioners (UNU, 2014).

2.2 Heavy metals and its sources in e-wastes

E-waste contains valuable ferrous and nonferrous materials such as copper, gold, platinum, steel, silver, aluminum, plastics, and glass that can be extracted and sold as secondary raw materials. However, some of these substances are toxic when they are released into the environment as a result of improper and uncontrolled recycling and dismantling methods. Table 1 lists some of the common heavy metals contained in various parts of e-waste, particularly heavy metals which are emitted to the environment during the illegal burning of e-waste.

Table 1 Heavy metals contained in e-waste

Substance	Applications
Cadmium (Cd)	Rechargeable Ni-Cd batteries, Phosphor emitters in CRT screens, Printer inks and toners, Switches, Connectors, Semiconductor chips, Printed circuit boards, Plastic stabilizer
Chromium VI (Cr)	Corrosion protection of untreated and galvanized steel plates, Data tapes, Floppy disks
Copper (Cu)	Conductivity of printed circuit boards, cathode ray tubes, connectors Electrical wiring
Lead (Pb)	Solder in printed circuit boards, Radiation shield in glass, Panels in cathode ray tubes (CRT) in computer monitors, Batteries
Mercury (Hg)	Cold cathode fluorescent lamps, Liquid crystal display (LCD) backlights, Alkaline batteries, thermostats, sensors, monitors
Nickel (Ni)	Rechargeable Ni-Cd batteries, Electron gun in CRT screens, Structural, printed circuit boards, cathode ray tubes
Zinc (Zn)	Anticorrosion coating, Cathode ray tubes
Arsenic (As)	Light-emitting diodes (LEDs), Liquid crystal display panels
Manganese (Mn)	Ni-Mn battery
Antimony (Sb)	Antimony trioxide as a flame-retardant additive in plastics and resins, as well as from use in electrical solders, Printed circuit boards
Beryllium (Be)	Silicon-controlled rectifiers (power supply) known to contain beryllium oxide

Source: Leung et al. (2006) and Brigden et al. (2005)

2.3 E-waste situation in Thailand

In Thailand, electronic equipment after being used by owners has been sold to repair/recycled junk shops. The dismantlers buy the e-waste from the retailed shops to separate valuable materials, and then sell them as secondary raw materials. The materials that cannot be sold are dumped and burnt in the area as called dumping site and burning site (Amphalop et al., 2020; Thongkaow et al., 2017; Vassanadumrongdee, 2014; Withayaanumas, 2017). There have been many well-known e-waste dismantling sites in Thailand that are located in (1) Khong Chai, Kalasin Province, (2) Khuang Nai, Ubon Ratchathani Province, (3) Daeng Yai and Ban Pao subdistrict, Buriram Province, which is the study area in this study, and (4) Sue Yai Utit, Bangkok.

2.3.1 E-waste volume growth in Thailand

According to ThaiPCD (2019), approximately 414,600 tons of e-waste were generated in 2018 which accounted for 65% of municipal hazardous waste. Of the 414,600 tons of e-waste generated in 2018, only 13% of e-waste was documented to be collected and recycled through proper practices, while 87% were managed and dismantled by uncontrolled e-waste dismantling activities, leading to the release of toxic substance contained in e-waste into the environment. The amount of e-waste generation in Thailand is expected to rapidly increase every year; by 2021, 13.42 million unwanted cell phones, and 3.65 million portable audio players are expected to be found. However, from the survey, amount and types of electronic equipment that was found most for e-waste dismantling sites of northeastern Thailand were televisions, washing machines, refrigerators, desktop computers, and fans because of more valuable materials (Thongkaow et al., 2017).

2.3.2 The procedure of informal e-waste dismantling

Typical informal activities have been operated in order to obtain valuable materials and precious metals contained in e-waste. The materials are then sold to the dealers in the area nearby. Unwanted parts of e-waste are dumped and burnt in the open fields of the villages. Thongkaow et al. (2017) and Vassanadumrongdee (2014) reported that the following manners that cause heavy metal contamination in environment are carried out in e-waste dismantling sites.

(1) Physical dismantling by using tools such as hammers, chisels, screwdrivers and bare hands to disassembly and smash electronic equipment
(2) Burning of wire cables to recover copper
(3) Desoldering components from printed circuit boards to retrieve copper
(4) Splitting compressor of air-conditioners and refrigerators to extract copper and ferrous
(5) Beating of cathode ray tube (CRT) screens to access metal plates and alloy plates

(6) Chipping plastics
(7) Disposal and open-burning of unwanted parts in fields (e.g. debris, chips, and foams)

2.4 Heavy metal contaminations in soils in the e-waste recycling and dismantling sites

The term “informal recycling” means the recycling activities that are carried out by people who resort to scavenging and waste picking for income and survival. The informal recycling of e-waste is usually done by primitive methods to retrieve valuable materials from e-waste, which primitive techniques allowing the emissions of toxic substances (Zhuang, 2019).

These informal and unregulated recycling methods of e-waste, without proper pollution control, could release substances into the environment via direct and indirect pathways, causing severe and harm to the local environment and workers. For instance, open burning of e-wastes to obtain valuable metals directly cause the contamination in soils. The release of particulate matter into the atmosphere from the e-waste open burning could indirectly contaminate the surface soil because of wet deposition (Leung, 2019).

Soil is a significant receptor of emissions from informal e-waste recycling. Hence, it is an important environmental media that can reflect the distribution and fate of the contaminants occurring in the environment. Since heavy metals are the main component of e-waste, persistent in the environment, and potentially accumulate in living organisms, many previous research have been studied heavy metal contaminations in soils in e-waste recycling site.

Leung et al. (2006) quantified the pollution levels generated from e-waste activities at Guiyu, Guangdong Province, China, and the impact on the environment. The soil samples were collected from the burnt plastic dump site, printer roller dump site, and reservoir. Cu, Pb, and Zn were the most abundant metals among the samples. Cu at the printer roller dump site (712 mg/kg), and the burnt plastic dump site (496 mg/kg) exceeded the Dutch action value of 190 mg/kg. Cd, Cu, Ni, Pb, and Zn contents at the printer roller dump site and the burnt plastic dump site exceeded the Dutch optimum values. For the reservoir soil, the heavy metal concentrations were below the limits for the natural background defined by the Chinese Environmental Quality Standards. This study concluded that the most seriously contaminated soil was in the burnt plastic and printer roller dump sites.

Ha et al. (2009) studied the contamination by trace elements at e-waste recycling sites in Bangalore, India. The soil samples were collected from (1) storage areas outside the main building of the e-waste recycling facility (BEF), (2) backyard recycling sites in the slum areas (BES), and (3) garden of the e-waste facility and rural areas in the suburbs of the city (BC). Cu and Mn were high in all soil samples, while

Hg was not detected in any of the soils from BEF and BC. Cu, Zn, In, Sn, Sb, Pb and Bi in BES and BEF were greater than those in BC. Cu, Zn, In, Sn, Pb in BES were higher than those in BEF, indicating that the crude method of recycling could elevate the levels of the trace elements to the environment than the controlled method in the licensed facility for e-waste recycling. Ag, Cd, Hg in BES were greater than those in BC; similarly, Hg in BES was higher than BEF as a result of the crude extraction methods of gold using Hg. It can be eventually concluded that e-waste processing activities and their disposal lead to elevated levels of some trace elements.

Luo et al. (2011) investigated the effect of uncontrolled e-waste recycling activities in Guangdong province of South China on the surrounding area. The soil was collected from vegetable gardens, paddy fields, incineration sites, deserted soil, and pond areas. The e-waste incineration sites revealed the highest concentrations of metals, with the average being 17.1 mg/kg of Cd, 11,140 mg/kg of Cu, 4,500 mg/kg of Pb and 3,690 mg/kg of Zn. This contamination was probably due to the burning of circuit boards and other metal chips in the recycling processes. These highly exceeded the action values of the Dutch standard elements and highlighted the significant impact of e-waste processing activities. The heavy metal concentrations in the soils collected from e-waste incineration site were higher than those from the pond area because e-waste combustion area is close to the ponds; they provided water for metal extraction processes. Thus, the heavy metals could be leached out from the burning sites to the ponds. In addition, the electrical debris was usually dumped beside ponds, and metals in these scraps could enter with rainwater into aquatic system. The concentration of paddy fields and vegetable gardens is a great concern because vegetables and rice were being grown in the soils close to e-waste recycling facilities. For agricultural soils, the soils were primarily contaminated by Cd and Cu, with the mean values more than three folds of the Chinese standard for agricultural soils, and the concentration exceeded the target value of the Dutch Standard. The study concluded that e-waste recycling activities were the dominant source of metal pollution in the vicinity. Elevated of Cu and other metals in the affected agricultural soils were consistent with the patterns observed at former e-waste open incineration sites, indicating that metals released from the processing of e-waste may enter the surrounding paddy fields and vegetable gardens via air deposition and water irrigation.

Isimekhai et al. (2017) assessed the contamination resulted from the recycling activities operated in an informal e-waste recycling site in Lagos State, Nigeria. The soil samples were taken from (1) the dismantling area where manual dismantling of electronics were conducted to recover metals like Cu, Al, and other precious metals, (2) the recycling area where open burning of some electronic components and wire cables were operated, and (3) the control site which was 500 m away from the study site. The study revealed that there was a wide range of concentration, and the descending order of the heavy metal in the study area was

Cu>Pb>Zn>Mn>Ni>Sb>Cr>Cd. Apparently, the total concentration of all heavy metals in the site exceeded those found in the control site, meaning that the e-waste recycling activities cause an impact of heavy metal contamination in the soils.

Oguri et al. (2018) collected the garden soil samples from the village of Bui Dau located in an e-waste processing area of Hung Yen Province in northern Vietnam. The median metal contents in the garden soil of Cu (771 mg/kg), Pb (580 mg/kg), and Zn (860 mg/kg) were up to 11, 5, and 4 times, respectively corresponding Vietnamese residential soil guidelines (70, 120, and 200 mg/kg for Cu, Pb, and Zn, respectively). The recycling activities operated in the study area were moderate-intensity activities, such as dismantling of metal and plastic and sorting of electrical parts. Metal contamination in the soil presumably originated from e-waste scraps, containing many heavy metals, that fell directly onto the soil surface. Moreover, the result showed that Cu and Sb contents in garden soil were positively correlated with the corresponding contents in floor dust. According to Xu et al. (2015), they have reported Pb, Sb, Cu, and Cd were emitted during e-waste recycling activities, so it can be assumed that the Cu and Sb contaminated in the soil derived from e-waste recycling activities carried out in the workshops.

In addition to the previous studies mentioned above, the heavy metal concentration in other studies at different e-waste sites and the soil guideline values are shown in Table 2 and Table 3, respectively.

Table 2 Mean metal contents (mg/kg) in soil reported in previous studies [a]

Location	Land use	Cd	Cu	Mn	Pb	Sb	Zn	Cr	Ni	Hg	As	References
Bui Dau, Vietnam	Simple household e-waste dismantling workshops	1.28	1,520	509	759	46	761	-	-	-	-	Oguri et al. (2018)
Bangalore, India[b]	Storage areas outside the main e-waste recycling facility (BEF) building	0.478	429	619	126	24	192	54	-	-	-	Ha et al. (2009)
	Backyard recycling sites in the slum areas (BES)	2.33	592	449	297	14	326	73	-	-	-	
	Rural areas in the suburbs of Bangalore city (BC)	0.165	22.8	390	22.8	0.43	41	57	-	-	-	
Guangdong Province, China	Vegetable garden	0.9±0.8	324±172	-	95.6±19.5	-	122±55.7	12.3±5.1	8.83±2.9	-	-	Luo et al. (2011)
	Paddy field	1.0±0.4	155±94	-	61.8±24	-	166±76.7	17.3±8.1	34.5±26.6	-	-	
	Incineration site	17.1±12.5	11,140 ±9,000	-	4,500±3,370	-	3,690±2,680	68.9±53	60.1±59	-	-	
	Pond area	5.45±7.43	3,550±5,480	-	1,880±3,340	-	1,160±1,580	38.9±38.4	32.2±20.7	-	-	
	Background of Guangdong province	0.06	17	-	36	-	47.3	50.5	14.4	-	-	
Guiyu, China	Burnt plastic dump site	1.7	496	-	104	-	258	28.6	155	-	-	Leung et al. (2006)
	Printer roller dump site	3.1	712	-	190	-	-	74.9	87.4	-	-	
	Reservoir	ND	9.2	-	55.4	-	78	3.4	8.4	-	-	
Lagos State, Nigeria (Dry season)	Recycling area	26.39 ± 2.57	3,277±277	115.3 ±8.86	2,418±289	38.46 ± 3.70	2,195±180	35.45 ± 3.59	40.80 ± 4.87	-	-	Isimekhai et al. (2017)
	Dismantling area	10.29 ±1.87	3,165±502	254.9 ± 24.4	911±111	22.51 ± 2.68	862.6±42.1	36.78 ± 2.68	77.4±10.7	-	-	
	Control area	0.87±0.02	14.7±0.08	1.88±0.02	20.62±0.14	0.22±0.01	24.53±0.86	0.3 ± 0.01	0.7 ± 0.02	-	-	

Location	Land use	Cd	Cu	Mn	Pb	Sb	Zn	Cr	Ni	Hg	As	References
Lagos State, Nigeria (Wet season)	Recycling area	12.69 ± 1.27	4,858± 318	92± 11.6	1,969± 157	35.36 ± 3.30	915± 42.7	23.04 ± 2.51	23.27 ± 2.54	-	-	Isimekhai et al. (2017)
	Dismantling area	8.67± 2.4	5,880± 636	120.8 ± 23.3	1,823± 230	58.4± 13.6	1,921± 200	49.6± 4.72	23.91 ± 2.3	-	-	
	Control area	0.45± 0.02	9.64± 0.06	1.26± 0.04	10.58± 0.07	0.15± 0.02	17.34± 0.45	0.19± 0.01	0.29± 0.01	-	-	
Wenling, China	Large-scale e-waste recycling plants (n=7)	3	180	-	180	-	343	101	49	0.5	-	Tang et al. (2010)
	Large-scale gold recovering plants (n=2)	5.1	424	-	144	-	203	90	114	1.8	-	
	simple household e-waste dismantling workshops (n=3)	4.4	327	-	313	-	299	77	64.3	221	-	
Guiyu, China	Acid leaching area	1.36	684.1	-	222.8	1706	572.8	7.43	278.4	-	-	Quan et al. (2015)
Guiyu, China	abandoned sites using acid processing	1.21	4,800	300	150	1,100	330	2,600	480	0.21	26	Jinhui et al. (2011)
Philippines	Formal site (Complied with environmental regulations)	0.3	350	800	53	-	140	-	16	-	-	Fujimori et al. (2012)
	Informal site (Not complied with environmental regulations)	2.9	810	900	650	-	1,000	-	64	-	-	
Thailand	Simple household e-waste dismantling workshops	-	58-12,986	30-715	40-4,556	-	158-4,258	-	12-183	-	-	Damrongsiri et al. (2016)

[a] Standard deviations are provided when available

[b] Geometric means

“-” means data is not available.

Table 3 Regulatory standards of heavy metals in residential and agricultural soil (mg/kg)

Standard	As	Cd	Cr	Cu	Mn	Ni	Pb	Zn	Reference
Chinese standard for agricultural soil	30	0.3	250	50	-	40	250	200	CEPA (1995)
Dutch soil quality standards									VROM (2000)
- Optimum value	29	0.8	100	36	-	35	85	140	
- Action/intervention value	55	12	380	190	-	210	530	720	
Thai standard for residential and agricultural soils	3.9	37	300	-	1,800	1,600	400	-	Thai Office of National Environment Board (2014)
England soil guideline values									DEFRA (2002)
- Residential soil									
- Agricultural soil	32	10	-	-	-	130	-	-	
	43	1.8	-	-	-	230	-	-	
Canada soil quality guidelines									C. E. P. A. CEPA (2007)
- Residential soil									
- Agricultural soil	12	10	64	63	-	45	140	250	
	12	1.4	64	63	-	45	70	250	
New Zealand soil standard									MFE (2011)
- Residential soil	17	3	460	-	-	-	210	-	
Screening value, residential of U.S. EPA	0.11	0.48	11	270	-	72	200	1,100	U.S. EPA (2001)

"-" means data is not available

2.5 Heavy metal fractionations in soils

The environmental mobility and bioavailability of heavy metals strongly depends on their chemical form or way of binding. Consequently, their biogeochemical pathways can only be studied on the basis of the determination of these forms. The determination of chemical forms is based on the use of leaching or extraction procedures. For heavy metals, the development and use of extraction schemes aimed to evaluate the metal fractions available to plants and environmentally accessible trace metals (Sahuquillo et al., 2007).

To analyze the heavy metal fractions, different analytical approaches are used (Lopez-Sanchez et al., 2002). The approaches based on extraction/leaching procedures are the most widely accepted and used. However, most of the work carried out focused on the use of sequential extraction as a tool to evaluate the availability of metals to plant or to study the metal distribution and/or mobility in polluted soils. To do so, several extracting reagents are applied sequentially to the sample according the following order: unbuffered salts, weak acids or buffered salts, reducing agents, oxidizing agents, and strong acids.

The fractions obtained when sequential extraction procedures are applied and related to exchangeable metals, metals mainly bound to carbonates, metals released in reducible conditions such as those bound to hydrous oxides of Fe and Mn, metals bound to oxidizable components such as organic matter and sulfides, and a residual fraction. Based on the modified standard sequential extraction method prepared by The European Community Bureau of Reference (BCR), there are main four geochemical phases as follows (Rauret et al., 1999):

(1) Acid-soluble phase: this phase is made up of exchangeable metals and others bound to carbonates that are easily dissolved in the environment, and readily mobile and available to plants or organisms.
(2) Reducible phase: this phase consists of metals bound to Fe and Mn oxides that can be released if the soil condition changes from the oxic to the anoxic state by the activity of microorganisms presenting in the soils. This form is potentially mobile and bioavailable.
(3) Oxidizable phase: this is metals bound to organic matter and sulphides which can be mobilized with time, oxidation or decomposition. This form is probably mobile and bioavailable.
(4) Residual phase: this is not usually available to biota as the metals are tightly bound within the structure of the soil, so it is stable and unavailable to biota.
(5) There are many fractionated extraction procedures.

Most of the published literature is based on the work of Tessier (Tessier et al., 1979), but a new approach, improvements and modifications are also proposed. In addition to Tessier's work, BCR procedure launched by the Community Bureau of

Reference, which is a harmonized sequential extraction procedure, is also widely used to determine metal partitioning in soils and sediments (Quevauviller et al., 1994; Rauret et al., 1999). To sum up, the brief details of these two methods are given in Table 4 and Table 5, respectively (Oyeyiola et al., 2011; Rao et al., 2008).

Table 4 Operating conditions required in the Tessier sequential extraction procedure

Stage	**Fraction**	**Reagents for 1 g of soil sample**	**Experimental conditions**
1	Exchangeable	8 mL of 1mol/L $MgCl_2$ (pH 7)	1 h at 25^{o}C
2	Associated with carbonates (acid-soluble)	8 mL of 1mol/L NaOAc (pH 5 with acetic acid)	5 h at 25^{0}C
3	Associated with Fe-Mn oxides (Reducible)	20 mL of 0.04 mol/L $NH_2OH \cdot HCl$ in 25% w/v HOAc (pH~2)	6 h at 96^{o}C
4	Associated with organic matter (oxidizable)	3 mL of 0.02 mol/L HNO_3 and 5 mL of 30% m/v H_2O_2	2 h at 85 ^{o}C
		+ 3 mL of 30% m/v H_2O_2	3 h at 85^{o}C
		+ 5 mL of 3.2 mol/L NH_4OAc	30 min at 25^{o}C
5	Residual	10 mL of concentrated HF/ 2 mL of concentrated $HClO_4$	Digested near dryness
		+1 mL of $HClO_4$/ 10 mL of HF	Evaporated to near dryness
		+ 1 mL of $HClO_4$	Evaporated until the appearance of white fumes

Source: Tessier et al. (1979)

Table 5 Operating conditions required in the modified BCR sequential extraction procedure

Stage	Fraction	Reagents for 1 g of soil sample	Nominal target phase	Experimental conditions
1	Exchangeable, acid soluble	40 mL of 0.11 mol/L CH_3COOH	Soil solution, exchangeable cations, carbonates	22±5^{o}C, 16 h, constant shaking (30±10rpm)
2	Reducible	40 mL of 0.5 mol/L $NH_2OH{\cdot}HCl$ pH 1.5 (with nitric acid)	Iron and manganese oxyhydroxides	22±5^{o}C, 16 h, constant shaking (30±10rpm)
3	Oxidizable	10 mL of 30% w/v H_2O_2	Organic matter and sulfides	22±5^{o}C, 1 h, occasional agitation +85°C, 1 h
		10 mL of 30% w/v H_2O_2		85°C, 1 h, Reduce the volume to a 1 mL (do not take to complete dryness)
		50 mL of 1 mol/L CH_3COONH_4 at pH 2		22±5^{o}C, 16 h, constant Shaking (30±10rpm)
4	Residual	8 mL HNO_3 and 4 mL HF and then 10 mL H_2O_2	Non-silicate minerals	22±5^{o}C , and then 250^{o}C

Source: Rauret et al. (1999)

2.5.1 Heavy metal fractionations studies in contaminated soils at e-waste sites

E-waste recycling or dismantling activities have caused serious heavy metal contaminations in soils. Since total concentration of heavy metals is inadequate to assess heavy metal mobility, bioavailability, and eco-toxicity in the e-waste area, sequential extraction has been studied to provide more information on the partitioning of the heavy metals by their associations with fractions. Many studies have assessed the potential bioavailability and mobility of heavy metals in e-waste recycling sites.

Zhao et al. (2011) investigated the chemical fractions of heavy metals in paddy fields. Soil samples (0-15 cm in depth) were collected from the rice production area of Wenling in southeast Zhejiang province, China, which is one of the well-known electronic and e-waste recycling centers. The samples were then analyzed for the total heavy metal concentration (Cd, Cu, Ni, Pb, and Zn) and metal fraction concentration using modified Tessier sequential extraction method providing five fractions. Since the carbonate bound fraction was not detected, the fractions of heavy metals included exchangeable, Fe-Mn oxide bound, organic bound, and the residual fraction. The paddy soils showed Cd contamination. The elevated levels Cd were predominantly in extractable fractions; Cd was largely partitioned with the exchangeable fraction with an average of 49.7%, followed by the Fe-Mn oxide fraction with an average of 37.2%. The percentage fractions of Cd decreased in the order of exchangeable > Fe-Mn oxide bound > residual > organic bound fraction. In contrast, soil Cu, Ni, Pb, and Zn were mainly in the residual (non-extractable) fractions with an average of 57.8%, 73.7%, 43.9%, and 70.8%, respectively. The fractions of Ni, Pb, and Zn followed the order of residual > Fe–Mn oxide bound > organic bound > exchangeable fraction; the fractions of Cu decreased in the order of residual > organic bound > Fe–Mn oxide bound fraction.

Luo et al. (2011) studied the effect of uncontrolled e-waste recycling activities on the surrounding environment of an e-waste site located in Guangdong province of South China. Soil samples were collected from e-waste former incineration sites and surrounding vegetable gardens, paddy fields, deserted area, and a reservoir area. Five fractions of heavy metals, Cd, Cr, Cu, Ni, Pb, and Zn, were investigated by using a modified Tessier sequential chemical extraction to evaluate the mobility and potential bioavailability of the heavy metals. The results revealed that Cu, Cr, Pb were predominantly associated with the residual fraction followed by the carbonate/specifically adsorbed phases, which accounted for more than 76%, 98%, and 75% of the total concentrations in soils. The high percentage of the exchangeable phase was found in the vegetable gardens and paddy fields than in those of the pond area and burning sites. For Ni and Zn in the soils of incineration site and pond area, they were mainly bound to the residual fraction, followed by the carbonate/specifically adsorbed phases. However, in the paddy field and vegetable soils, the two metals in the exchangeable fraction exceeded those in the carbonate/specifically adsorbed phases. Higher concentrations of metals in the exchangeable phase would indicate high solubility and bioavailability, meaning that the metals can be more readily taken up by plants grown in soil.

Luo et al. (2011) also reported Cd showed a distinct character in the paddy field and vegetable soils, where the extractable form accounted for more than 70% of the total concentrations. The second dominant form of Cd in the paddy field and vegetable soils was the carbonate/specifically adsorbed phase, accounting for more than 10% of the total Cd. By contrast, in the incineration site and pond area, Cd was mainly associated with the residual fraction, which accounted for 66% and 94% of the total Cd, followed by the carbonate/specifically adsorbed phase with 16% and 3% of the total in soils.

Isimekhai et al. (2017) evaluated heavy metal contamination and pollution due to the recycling activities in Lagos State, Nigeria. Soil samples were collected from the dismantling site where manual dismantling, breaking up, and sorting of e-waste were performed, and the recycling site where workers operated open burning of e-waste especially wire cables, and recovering precious metals, such as Cu and Al. The soil samples were collected in both dry and wet seasons. The three-step sequential extraction method was operated to study bioavailability of heavy metals in soils. The resulted showed the same trend for dry and wet season. The order of percentage association of metals with the exchangeable form, which represents the readily available fraction, in descending order as Cd>Sb>Mn>Zn>Cu>Ni>Pb>Cr. Cd followed by Sb is potentially the most bioavailable heavy metal in the soils, whereas Cr is the least available for uptake by plants. The order of metals associated with organic matter fraction is Pb>Sb>Zn>Cu>Cd>Mn>Ni>Cr. In the residual fraction, the order of association with the alumina-silicate minerals is Cr>Ni>Mn>Cu>Zn>Pb>Sb>Cd. The results implied that Cd is potentially the most bioavailable heavy metal, posing a risk to environment as it is associated with the exchangeable fraction.

He et al. (2017) aimed to evaluate the soil heavy metal pollution in the e-waste recycling areas located in Qingyuan and Guiyu of south China. They adopted the modified BCR sequential extraction procedure to determine the heavy metal fractionation in the soil (0-10 cm in depth) collected from the e-waste area. The results from Qingyuan showed that most Cd resided in the acid soluble fraction (63.6%) and the reducible fraction (20.6%), meaning that Cd could pose significant ready and potential risk to the environment and ecosystem. A significant portion (>20%) of Zn, Ni and Cu were presented in the acid soluble fraction. Cu (39.4%) and Pb (45.5%) partitioned mainly in the reducible fraction indicating the potentially mobile and bioavailable in soils. In contrast, Cr (84.6%), As (76.6%), Ni (64.7%) and Zn (63.4%) were primarily presented in the residual fraction, which means that they were not available for biogeochemical cycling. Regarding the fraction of heavy metals in the soils from Guiyu, 60.1% of Cd was presented in the acid-soluble fraction, while most of Cr (85.1%), Zn (72.7%), Ni (65.0%), As (49.6%) and Cu (38.9%) were partitioned in the residual fraction. Pb (59.7%), As (34.8%) and Cu (30.2%) had significant portions distributed in the reducible fraction causing high potential mobility and bioavailability. Overall, Cd had much higher mobility and bioavailability compared to the other heavy metals (Cu, Cr, Pb, Ni, and Zn) in the surface soils impacted by e-waste recycling activities.

2.6 Factors affecting the heavy metals in soils

2.6.1 Climatic condition

The heavy metal concentration levels in soils have been reported to be affected by season changing and depth of soils (Isimekhai et al., 2017; Olafisoye et al., 2013). Olafisoye et al. (2013) collected the soil samples from the e-waste dumpsite in Nigeria at 0-15 cm and 15-30 for both wet and dry season. The results showed that the heavy metals (Pb, Cd, Zn, Cr, and Ni) in the dry season were higher than those in the wet season; this can be attributed to the leaching of cations down the profile by rainfall. Isimekhai et al. (2017) also reported that the concentration of Cd, Cu, Mn, Ni and Zn in the dry season were significantly higher than those in the wet season ($p < 0.001$). The precipitations result in displacement of ions in the soils that can increase leaching of basic ions (Isimekhai et al., 2017).

2.6.2 Soil depths

Regarding the soil depths, Olafisoye et al. (2013) reported the total heavy metal concentration decreased with depth in soil samples. The concentrations of the heavy metals in the surface layer (0-15 cm) were found to be higher than those in sub-surface layer (15-30 cm). It can be discussed that the sub-surface soil is less considerably influenced by the anthropogenic supply, which was the informal e-waste recycling activity in the area, than the surface soil. The result is in accordance with earlier study (Adelekan & Alawode, 2011). Amphalop et al. (2017) collected soils from an informal e-waste dismantling site, and also found that the Cu and Pb concentrations in the surface layer (153 mg/kg of Cu and 6.69 mg/kg of Pb) were higher than in the sub-surface soil (21.73 mg/kg of Cu and 1.84 mg/kg of Pb).

2.6.3 Soil properties affecting the fate and transport of heavy metals in soils

Heavy metals are distributed in soils and associated with them in several ways, including adsorption, ion exchange, precipitation, and complexation. Changes in environmental conditions, such as pH, temperature, redox potential, and organic ligand concentration, are able to cause metals released from solid to liquid phase (Rieuwerts et al., 1998).

Natural conditions or soil properties can provide information on the identification of the main binding sites of heavy metals with certain soil and the phase associations of heavy metals in soils. For example, changes in the ionic composition can affect adsorption-desorption reactions, and a decrease in pH could cause the release of metals that are weakly retained on soil matrix or metals co-precipitated with carbonates. This soil condition is related to exchangeable and acid soluble forms (Sarkar et al., 2014).

Decreasing the redox potential can result in the dissolution of oxides, unstable under reducing conditions, which is related to reducible forms. Changes in oxidizing conditions may lead the degradation of organic matter and release of complexed metals that directly associate to oxidizable forms. Lastly, the destruction of primary and secondary mineral lattice releases heavy metal retained within the crystal structure, which refers to the residual forms. The soil properties governing the partition of metals between the solid and solution phase of soils are summarized as follows:

2.6.3.1 pH

pH is master variables for the geochemical processes controlling the solubility of heavy metals. Heavy metal cations tend to desorb or dissolve from solids as pH decrease. They adsorb or precipitate as pH increase, but only up to a certain limit after which they are again solubilized (Rieuwerts et al., 1998).

Similarly, metal solubility tends to increase at lower pH and decrease at higher pH. The association between adsorption and pH is due to competition of H^+ ions for adsorption sites at low pH resulting in decreased metal adsorption. However, the competition of H^+ ions at low pH may be reduced when metal concentrations are relatively low due to specific adsorption. In sequential extraction studies, a decrease in soil pH generally results in an increase of the water-soluble or exchangeable metal forms. For example, Dudka et al. (1996) revealed a negative correlation between soil pH and water-soluble Zn and exchangeable Cu. Likely, Xian and Shokohifard (1989) observed that a decrease in soil pH from 7 to 4.55 increased exchangeable levels of Pb, Cd, and Zn.

2.6.3.2 Redox potential

Redox is the reduction-oxidation reaction that reflects the processes involved in the flow of electrons from the reducing agents (reducer) to an oxidizing agent (oxidant). The abundance of electrons in a soil environment can be quantitatively expressed in terms of electrode potential (E_h) or electron activity. In a soil system, redox couples may be present: Fe^{2+}-Fe^{3+}, Mn^{2+}-MnO_2, NO_3^--NO_2^--NH_4^+, H_2S-SO_4^{2-}, and CH_4-CO_2. Manganese oxides also accumulate in soils by the specific adsorption of Mn^{2+} by freshly formed or “seed” Mn oxide surfaces. The adsorbed Mn^{2+} is then readily oxidized by O_2, increasing the volume and the surface area of the Mn oxide precipitate.

Redox zones in soils indicate the ranges of E_h controlled by various redox couples. *Oxidized soils* are defined as ($E_h > 414$ mV; O_2 present); *moderate reduced soils* ($120 < E_h < 414$ mV; O_2 absent, $NO_3^- \leftrightarrow N_2$ and $MnO_2 \leftrightarrow Mn^{2+}$); *reduced soils* ($-120<E_h<120$; $Fe^{3+} \leftrightarrow Fe^{2+}$); *highly reduced soils* ($E_h < -120$; $SO_4^{2-} \leftrightarrow S^{2-}$) (Essington, 2015). According to Evans (1989), soil in dry season typically shows high redox potentials because of the well aerated soils, while soils prone to waterlogging is likely to have low Eh values.

In the *oxic* region ($E_h > 414$ mV at pH 7), redox status is primarily controlled by the redox reactions of oxygen, and then nitrogen when oxygen has been depleted. Manganese and Fe control the redox processes in the *suboxic* region (120 mV < E_h < 414 mV at pH 7). For the *anoxic* region, the redox reaction of sulfur control redox chemistry when potentials are below 120 mV at pH 7 (Essington, 2015).

Under oxidizing and slightly reducing conditions, the solubility of heavy metal cations decreased due to the precipitation of oxides, hydroxides or carbonates. Under reducing conditions, heavy metal cations and metalloid can be removed from solution as sulfide minerals if sufficient sulfur is available. If not, reduction implies the loss of binding capacity by Fe and Mn hydroxides for cation and consequently induces contaminant mobility and bioavailability. The effects of redox condition on metal solubility have been studied by many studies. Being reported in Chuan et al. (1996), when solubility was compared under the same pH values, it was observed that metal solubility increased as redox potential decreased, and heavy metals are more readily dissolved in waterlogged soils. Rieuwerts et al. (1998) revealed that Zn availability increase in poorly drained, waterlogged soils, and Pb dissolved by acetic acid were higher in soils of impeded drainage (1.9 µg/g) than in freely drained soil (<0.1 µg/g). Chuan et al. (1996) reported the solubility of Zn, Cd, and Pb in soils increased when redox potential decreased. It can be explained by the dissolution of Fe-Mn oxyhydroxides under reducing conditions resulting in the release of adsorbed metal.

In contrast to the above findings, some authors reported that waterlogged and low Eh conditions do not favor an increase in metal solubility. Xiong and Lu (1993) found that waterlogging of soil will considerably decrease Cd bioavailability due to the formation of cadmium sulphide (CdS) and will also decrease water soluble Zn. Sims and Patrick (1978) reported that water soluble Zn decreased at low Eh.

2.6.3.3 Organic matter content

Organic matter (OM) mainly comes from decomposing plant materials and animal including cells and tissues of soil organisms, and substances exuded from plant roots and soil microbes. OM content is usually higher in surface soils or the root zone, and it decreases along the depths in the soil profiles (Fairbrother et al., 2007). Generally, the organic functional group present in OM has a high affinity for metal cations; heavy metals adsorbed electrostatically on the negative charges formed on the OM in soils, indicating the adsorption behavior of metals cation in soils.

With respect to the adsorption behavior, OM is a contributor to the ability of soil for retaining heavy metals in soils. Zeng et al. (2011) reported that the majority of metals in soils were bound to OM, and the metals are unavailable to plants. Heavy metal adsorption onto soil constituents, in addition, declined with decreased OM in soils. The result also showed that Pb, Zn concentration in grains were significantly negatively correlated with OM content in the soils. Similarly, Sungur et al. (2014) also revealed similar result that Cd is held tightly by OM because of the adsorption behavior.

However, OM also supplies organic chemicals to the soil solution that can serve as chelates, which are soluble organic compounds that bind metals such as copper, iron, manganese, and zinc, and increase their solubility and availability to plants (McCauley et al., 2009). For instance, Zeng et al. (2011) found the relationship between OM and heavy metals, i.e., Cu, Fe, and Zn, in straw were significantly positively correlated with OM. This is because the dissolved OM in soil could enhance the mobility and uptake of heavy metals to plant roots.

2.6.3.4 Soil texture

Soil texture can be divided into sand, silt, and clay fractions. This term is defined by particle size fractions of the soil with respective classification of > 50 µm, 2-50 µm, and <2µm. The soil texture represents the measured distribution of particle sizes, and the relative proportions of the various size ranges of particles in soil. The widely used soil particle size classification developed by the U.S. Department of Agricultural (USDA) is graphically presented in Figure 2. Soil texture class names are determined by the relative mass percentages of sand, silt, and clay-sized particles in the soil; there are 12 soil texture classes (Yolcubal et al., 2004).

Among three fractions, higher surface area for adsorption, presence of clay minerals, organic matter, Fe-Mn oxides, and sulphides are naturally occurred in clay fraction. In term of adsorption, the strong affinity of metals to the soil fraction is demonstrated by the order of clay> silt> sand, indicating that the amount of clay fraction in soils could influence the immobilization in soils (Andersson, 1979).

Several studies report the importance of clays in the immobilization of metals. Sungur et al. (2014) found the positive correlation between the clay content of soil and the residual fraction of Mn and Zn. In addition, it was reported that the amount of adsorbed Cd is increased along with an elevated amount of clay. This can be suggested that an increase in soil clay content may decrease the mobility and bioavailability of metals. Moreover, Qian et al. (1996) studied the effects of the particle-size distribution on the concentration of heavy metals in soil. It was reported that the finer particles, clay particles, show higher concentration of metals, and the extractable amounts of Co and Pb were also high in the clay fraction. High accumulation of metal in the clay fraction is resulted from the high surface area of clay for adsorption and from the presence of clay minerals, organic matter, Fe-Mn oxides and sulphides. The metals are probably adsorbed on clay minerals or are blocked in the clay lattice of soils.

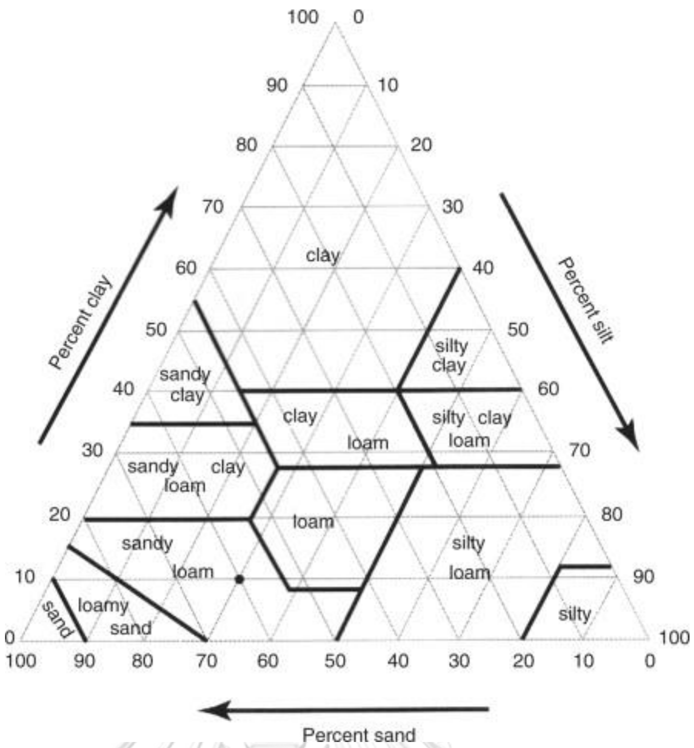


Figure 2 Soil texture triangle based on USDA particle-size classification (Yolcubal et al., 2004)

2.7 Ecological risk assessment in soils

2.7.1 Potential Ecological risk assessment

Since accumulation of metals can decrease soil quality, a comprehensive investigation of the status and ecological risk of heavy metal contaminated soil in the e-waste dismantling site are crucial for control of the long term soil pollution.

The potential ecological risk index (PERI) defined by Hakanson (1980) that focuses on heavy metals' abundance and their toxicological effects has been adopted to provide a simply quantitative value on the ecological risk caused by heavy metal contamination in soil (Han et al., 2019; He et al., 2017; Isimekhai et al., 2017; W. Wu et al., 2018; W. Zhao et al., 2015). The degrees of the ecological risk factors of the individual heavy metals (Er) and PERI is categorized as mentioned in chapter 3 (Hakanson, 1980). PERI is the summation of the ecological risk factors of the individual heavy metals (Er), and the risk indices can be calculated as mentioned in chapter 3.

According to ecological risk studies based on Hakanson's method in e-waste recycling sites, in Guiyu area, the largest e-waste recycling center in China, ecological risk of As, Cr, Cu, Ni, Co, Zn, Pb, and Mn were all less than 40, indicating that these heavy metals posed a low ecological risk in the soil. In contrast, Cd posed a moderate ecological risk (He et al., 2017; W. Zhao et al., 2015).

Similarly, in Qingyuan area, another e-waste recycling is in China, He et al. (2017) studied the risk posed by seven heavy metals and reported that the ecological risk of Cd posed the very high (Er = 269.91) and the PERI was in the category of considerable. Conversely, Han et al. (2019) also studied the risk in Qingyuan revealed that Cd and Sb posed extreme ecological risk, and PERI obtained from nine heavy metals was greater than 600, indicating a very high potential ecological risk.

Regarding the ecological risk at the e-waste recycling site in Nigeria, Isimekhai et al. (2017) reported similar results to above studies that the PERI in the study site indicated an extreme potential ecological risk that mainly attributable to the high levels of Cu followed by Cd and Pb.

2.7.2 Risk Assessment Code

The total contents of heavy metals cannot provide sufficient information on forms, mobility, bioavailability, or potential risks to the environment, therefore, the fractionation analysis of metals in soils is useful to assess the heavy metal contamination, especially accumulate in environment as an effective indicator of ecosystem quality which was introduced the certain code namely RAC or risk assessment code by Perin et al. (1985).

The RAC has been applied to evaluate the availability of metals in soils and then determine the environment risk of metals; RAC mainly compares the sum of the exchangeable and carbonate fractions with the total extracted for evaluating the availability of metals in soils. These fractions are considered to be weakly bonded metals that equilibrate with the aqueous phase and then become more rapidly bioavailable. RAC can be calculated as follows:

$$RAC = 100 \times (F_1) / \text{Total heavy metal concentration}$$

Where;

F_1 = exchangeable fraction and weak acid soluble

The risk assessment code has been classified into five risk levels according to Perin et al. (1985) as mentioned in chapter 3.

2.8 Soil contamination assessment

2.8.1 Enrichment factor (EF)

Enrichment factor has been applied in many previous studies to assess the intensity of metal contaminations resulted from anthropogenic activities to those from the natural sources. The calculation of EF is normalized by reference metals, like Al, Fe, Mn, and Rb, which are particularly stable and soils and naturally abundant in soils. Consequently, the EF can indicate different levels of metals enrichment in soils. EF values for metals being lower than 2 suggests that the metals concentration could be a result of natural soil weathering processes. However, EF for metals being higher than 2 indicates that the metal abundance in soils could be contributed to non-parent materials in soils or anthropogenic activities (Barbieri, 2016; Khalilova & Mammadov, 2016).

W. Zhao et al. (2015) aimed to assess the anthropogenic activities on the trace element concentrations at a typical e-waste recycling area in southeast China. The EF showed moderate enrichment level in the e-waste recycling site of Hg, Co, Ni, Zn, Sn, Li, and Be, indicating that the enrichment of these metals could be derived from the e-waste recycling activities carried out in the site. In contrast, they found that the As enrichment was in the minimal level as the EF value being lower than 2, so it could be concluded that As in various soil samples was influenced by a metal source of crustal soil rather than anthropogenic sources. Regarding the results of EF for metals in soil among different soil layers, the EF of Ni, Cu, Hg, Cd, Pb, and Sn was decreased from surface to subsurface layers of soil. Therefore, the result showed the retention of metals in the surface layer of soils, and also showed the limit of metal mobility in soils.

Pradhan and Kumar (2014) studied the enrichment level of metals in the e-waste recycling in India. Enrichment factor was applied, and the results showed that EFs values of As, Cd, Cu, Pb, and Zn were categorized in extremely high enrichment level in the e-waste recycling site, and e-waste dumping site, which is the highest level of metal enrichment. Comparing the EF values in the e-waste recycling sites to the surrounding residential areas, the results revealed that the occurrence of heavy metals in soil taken from the e-waste sites were extremely high compared to the that in soils collected from the residential areas. This result could confirm that the metal contamination in the study area was because of e-waste recycling activities.

2.8.2 Principal component analysis (PCA)

As there have been many metals studied in some certain studies, principal component analysis (PCA) has been utilized to reduce many metals into sets of associated metals based on metal concentration in studies. The sets could be either natural source-derived metals or anthropogenic-derived metals. Thus, PCA explains the possible sources of metals in an environmental media; metals being in the same principal component (PC) indicate that they are from the same pollution sources. He et al. (2017) studied the occurrence and distribution pattern of heavy metals in an e-waste recycling site and industrial zone in China. After the PCA was applied, there were 5 principal components showing 5 groups of associated metals in the site. With more than 90% of the accumulative variance, it suggested that these 5 principal components represented the studied metals effectively. PC1 was dominated by Cu, and Cd with the factor loading of 0.75, and 0.91, respectively. Furthermore, PC2 was dominated by Pb, and Zn with the loading of 0.93, and 0.71, respectively. The results presented in PC1 and PC2 indicated that Zn, Pb, Cu, and Cd primarily come from the anthropogenic sources. PC3, PC4, and PC5 comprised of As, Cr, and Ni, respectively. These metals were reported to be originated from soil parent materials.

In addition, in Han et al. (2019) study, the total concentration of metal contents were imported to PCA in order to identify the probable source of the metals in a e-waste recycling area in China. The PCA reported that the PCA represent the metals in the study area well due to the 70.7% of total variance. There were 2 principal components reported in the study; Cd, Cu, Hg, Mn, Ni, Pb, and Sb were in the first principal components which suggested that they were from the same origin. As the mentioned metals in the first principal have been used in electronic products to give aimed properties in the equipment, it could be concluded that metals contained in PC1 were from the e-waste related activities. In contrast, the second principal could represent the natural sources of metals and other types of human activities leading to the contamination of metals in soils. However, they also reported that different pattern of e-waste dismantling or recycling activities could contribute to different metal pollution patterns in soils.

Similarly, PCA was also used to identify or group the associated metals in dust generated from e-waste recycling activities in the Philippines. The PCA represent 2 different groups of the studied metals. The PC1 composed of Fe and Mn, which were metals derived from crustal soils. PC2 composed of Cu, Pb, Ag, Cd, Zn, Co, Ni, and As; these group of the metals could represent an impact of metal pollution. Yuan et al. (2014) focused the contents of metal in soils in the urban area, and PCA was used to identify the potential origin of metals in the soils. In PC1, Pb, Cu, Cd, Hg, and Zn were dominated in the PC. With high coefficient of variation (CV) and high enrichment factor, the results implied that these elements might be originated from the same pollution source, and reflected the anthropogenic origin in the urban soils. Because various types of factories located in the urban area has discharged Cd, Cu, Pb, and Zn to the environment through combustion of coal, and automobile exhaust, the presence of the mentioned elements could be the result of industrial or anthropogenic activities. PC2 comprised of As, Cr, Ni, and V, and the mean concentrations of these elements were comparable with the background values. The occurrence of these elements could be dominated by soil parent materials and natural processes.

Chapter 3

Materials and methods

3.1 Study area

The study area is located in two e-waste dismantling villages. The first village is in Daeng Yai (DY) subdistrict, Ban Mai Chaiphot district, and the second village is in Ban Pao (BP) subdistrict, Phutthaisong district, Buri Ram province. There have been 105 households in Daeng Yai subdistrict (DY) and 68 households in Ban Pao subdistrict (BP) involved in uncontrolled/unauthorized e-waste handling for more than ten years.

E-waste dismantling activities, such as manually dismantling of e-waste to recover the precious metals and dumping of unwanted materials in backyards, are performed in the villages where both e-waste (EW) and non-e-waste dismantling houses (NE) are located. Nearby the e-waste dismantling houses, groundwater well (GW) where the villagers usually rely on for irrigation and drinking, are located. In GW area, the soils were collected from the surrounding area of the well. In addition, open burning sites (OB), where burning of e-waste has been carried out, are located among paddy fields (PF) in the villages, and electronic debris is also disposed in the open burning sites. Table 1 presents the soil sampling sites classified into 3 categories including control site, which is located in northwestern of study area over 5 km distance. The location of sampling sites of 2 subdistricts are illustrated in the maps as presented in Figure 3 and Figure 4, respectively. In addition, the pictures of the soil sampling points were showed in appendix A.

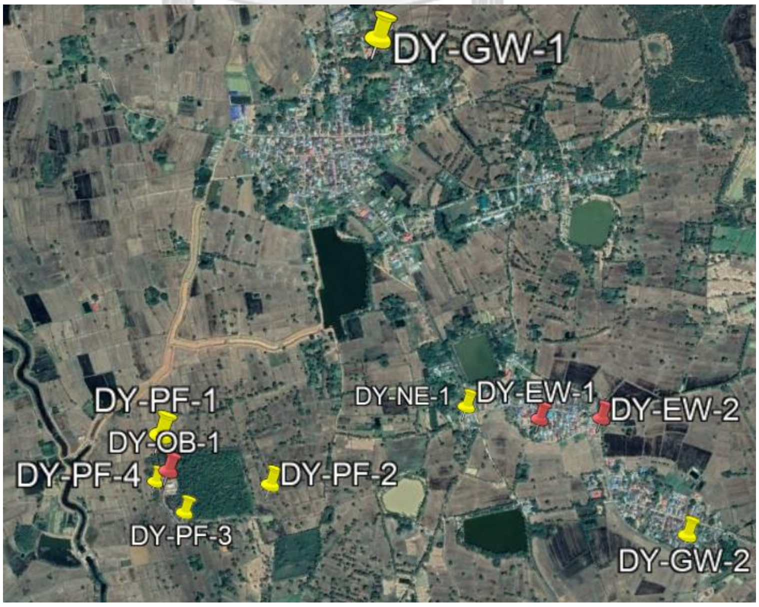


Figure 3 Location of sampling sites in DaengYai subdistrict, Buriram Province, Thailand; red pins refer to potential sources (OB and EW); yellow pins refer to potentially affected area (PF, NE, and GW)

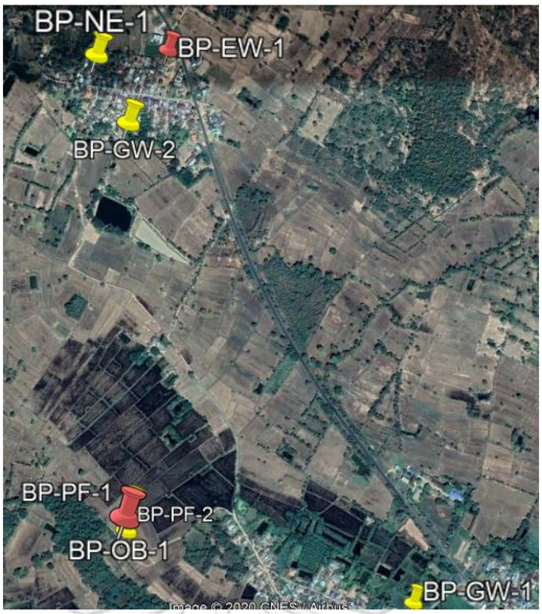


Figure 4 Location of sampling sites in BanPao subdistrict, Buriram Province, Thailand; red pins refer to potential sources (OB and EW); yellow pins refer to potentially affected areas (PF, NE, and GW)

3.2 Soil sampling

For each sampling site, a 2mx2m quadrat was made, and the depth intervals of 0-15 cm. (surface soil) and 15-30 cm. (sub-surface soil) were divided (Figure 5). For each depth, three sub-sample soils were collected diagonally by a shovel and then mixed to provide a kilogram of composite soil, and were kept in clean polyethylene bags. To avoid cross contamination, the shovel was carefully clean with deionized water, and then make it dry by dried wipes prior to sampling study soils. The soil sampling was conducted in both dry and wet season. Therefore, there were 72 soil samples in the study. A geo-referenced coordinates sampling location was recorded using a handheld GPS.

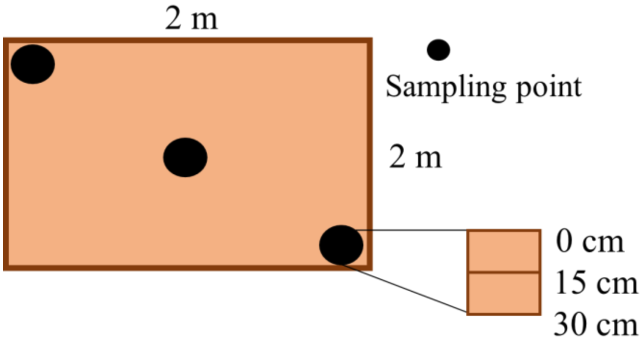


Figure 5 Soil sampling at each sampling site

Table 6 Description of sampling points

Categories	Sampling sites	Description	Sampling codes and GPS coordinates	
			Daeng Yai (DY)	BanPao (BP)
Potential sources	Burning site (OB)	Open burning of wires is performed to recover copper, and the sites serve for disposal of unwanted parts.	DY-OB-1 (15°34'48.59"N 102°52'47.06"E) (approximately 6,270 m^2)	BP-OB-1 (15°36'34.77"N 102°54'8.04"E) (approximately 2,000 m^2)
	E-waste dismantling house (EW)	This site is the house served as a storage area of e-waste; physical dismantling of e-waste is operated to recover the precious metals, and unwanted materials of e-waste are dumped in household areas.	DY-EW-1 (15°34'47.27"N 102°53'28.98"E) DY-EW-2 (15°34'46.18"N 102°53'35.76"E)	BP-EW-1 (15°37'16.10"N 102°54'12.20"E)
Potentially affected area	Paddy fields (PF)	This site is located beside the burning site, and rice cultivation is taken place in the paddy fields.	DY-PF-1 (15°34'52.14"N 102°52'46.71"E) DY-PF-2 (15°34'45.21"N 102°52'57.99"E) DY-PF-3 (15°34'43.97"N 102°52'48.07"E) DY-PF-4 (15°34'47.93"N 102°52'45.53"E)	BP-PF-1 (15°36'36.40"N 102°54'8.72"E) BP-PF-2 (15°36'34.16"N 102°54'8.36"E)
	Non-e-waste dismantling house (NE)	This site is the house of non-e-waste dismantling people which located among the e-waste dismantling houses.	DY-NE-1 (15°34'49.94"N 102°53'21.18"E)	BP-NE-1 (15°37'15.36"N 102°54'5.67"E)
	Ground water well area (GW)	This site is in the residences of the local people in village.	DY-GW-1 (15°35'29.84"N 102°53'18.23"E) DY-GW-2 (15°34'32.04"N 102°53'42.90"E)	BP-GW-1 (15°36'28.11"N 102°54'34.26"E) BP-GW-2 (15°37'9.45"N 102°54'8.51"E)
Control area	Control area	This site is located approximately 5 km from the village.	Control (15°36'26.54"N 102°51'44.73"E)	
Reference site (ref)	Reference	forest and uncultivated land in the village located approximately 5 km from the village	Ref (15°35'47.30"N 102°51'11.56"E)	

3.3 Soil analysis

3.3.1 Soil preparation prior to chemical analysis

All soil samples were air-dried at room temperature after manual removal of stones and other large materials, and grounded with a porcelain mortar and pestle. The grounded soils were passed through 2 mm mesh sieve, and then collected in polyethylene bag in desiccator for further analysis.

3.3.2 Soil physico-chemical analysis

The soil properties were studied for 4 parameters, which were soil pH, oxidation-reduction potential (ORP), soil organic matter (SOM), and soil texture. These parameters were analyzed by the method as shown in Table 7.

Table 7 Analytical methods for soil physico-chemical analysis

Soil properties	Methods
Soil pH	1:2 soil/deionized water (DI water) suspensions (Gleason et al., 2003; ThaiLDD, 2010).
ORP	
Soil texture	Hydrometer method (Bouyoucos, 1962)
SOM	Walkley-Black method, (Walkley & Black, 1934)

Soil pH and ORP were determined using a HACH sension3 pH meter and a HACH sension156 ORP meter in the ratio 1:2 (wt/vol).

For the hydrometer method, briefly, 50 g of soils were weighted and soaked with Calgon (sodium hexametaphosphate and sodium bicarbonate) overnight. 1000 mL of deionized water were added to the soaked soil in a 1000 mL cylinder, and then were agitated to make the dispersion. The amount of each particle group (sand, silt, clay) were determined using a hydrometer. At the end of 40 seconds, 2 hours, the hydrometer was inserted to obtain the hydrometer readings, and the temperature of the soil mixture were also be noted. The hydrometer reading provided grams of soil in suspension per liter of solution. The blank cylinder without soil was made following the same procedures. According to the principal of Stokes' law, the soil particles fall out of suspension at different rates over time based on particle size, therefore, the hydrometer reading was able to determine the amount of each particle size present in the soils used to determine the soil texture class by using a soil textural triangle.

To determine SOM, approximately 0.5 g was weighed, 10 mL of standard potassium dichromate solution added, and swirled to mix. 15 mL of concentrated sulphuric acid were added gently and mixed. The flasks were allowed to stand for 30 minutes. Five drops of ferroin were added and the resulting mixtures were titrated with ferrous ammonium sulphate until color change from blue green to violet red was observed. Total organic carbon was determined using an appropriate mathematical expression and multiplied by a factor to obtain the total organic matter.

3.3.3 Total heavy metal analysis

All glass wares were soaked overnight by a 10% HNO_3 acid solution and then rinsed with DI water. Approximately 0.5 grams of the sieved sample was weighed into a microwave vessel and then 12mL of aqua regia (9mL of 37% HCl : 3mL of 65% HNO_3) were added following U.S. EPA method 3050b (EPA, 1996). The sample was digested following the temperature program in Table 8. The digested samples were filtered through a filter paper no.42 (pore size 2.5 µm) into 25mL-flasks and were adjusted the volume by DI water and stored in polyethylene bottles prior to instrument analysis. The digested samples were analyzed for As, Cd, Cr, Cu, Mn, Ni, Pb, and Zn using inductively coupled plasma-optical emission spectroscopy (ICP-OES) (Figure 6). The soil samples were analyzed in triplicate for quality control. Blank determination was carried out without soil samples. The final concentrations of the heavy metals in soil samples were calculated using the equation 3.1.

$$Heavy\ metal\ in\ soils\ \left(\frac{mg}{kg\ soil}\right) = \frac{A\ x\ B}{1000\ x\ C} \tag{3.1}$$

Where;

A is the heavy metal concentration (µg/L)

B is the final volume of 25 mL (mL)

C is the soil weight (g)

Table 8 Temperature gradient used for the microwave assisted digestion

Step	Time	Temperature
1	5minutes	100 ^{o}C
2	15minutes	200 ^{o}C
3	15minutes	200 ^{o}C

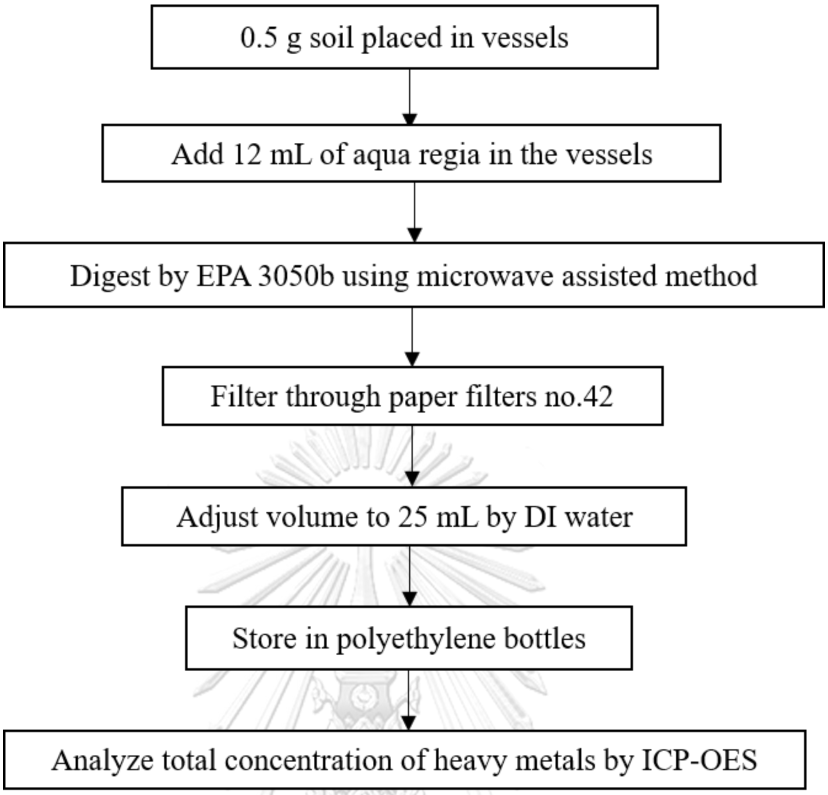


Figure 6 Soil digestion procedure in this study

3.3.4 Sequential extraction method

To assess the mobility and potential bioavailability of the heavy metals, sequential extraction was conducted based on the modified BCR sequential extraction. Approximately 1.0 g of soil sample was extracted sequentially into the exchangeable/acid soluble (step 1 - CH_3COOH) (F1), reducible (step 2 -$NH_2OH.HCl$) (F2), oxidizable (step 3 - H_2O_2/CH_3COONH_4) (F3), and Residual (step 4 – HNO_3/HF) (F4). In this study, only the first three BCR steps were obtained because of their potential bioavailability. The experimental conditions of the modified BCR sequential extraction are shown in

Table 5 and Figure 7.

At each step, the samples were centrifuged for 20 minutes at 3000 rpm in room temperature, and the supernatant was collected with a pipette. The obtained supernatant was stored in polyethylene bottles for further analysis. The residue was later washed by 20 mL DI water. After centrifuging for 20 minutes at 3000 rpm, the washed water was separated. The residue was taken to the next step. However, the study soils were determined in first three fractions to study the availability in environment and the effects on ecological risk of each heavy metal. The contents of As, Cd, Cr, Cu, Mn, Ni, Pb, and Zn in the obtained extracts after each step were determined using ICP-OES.

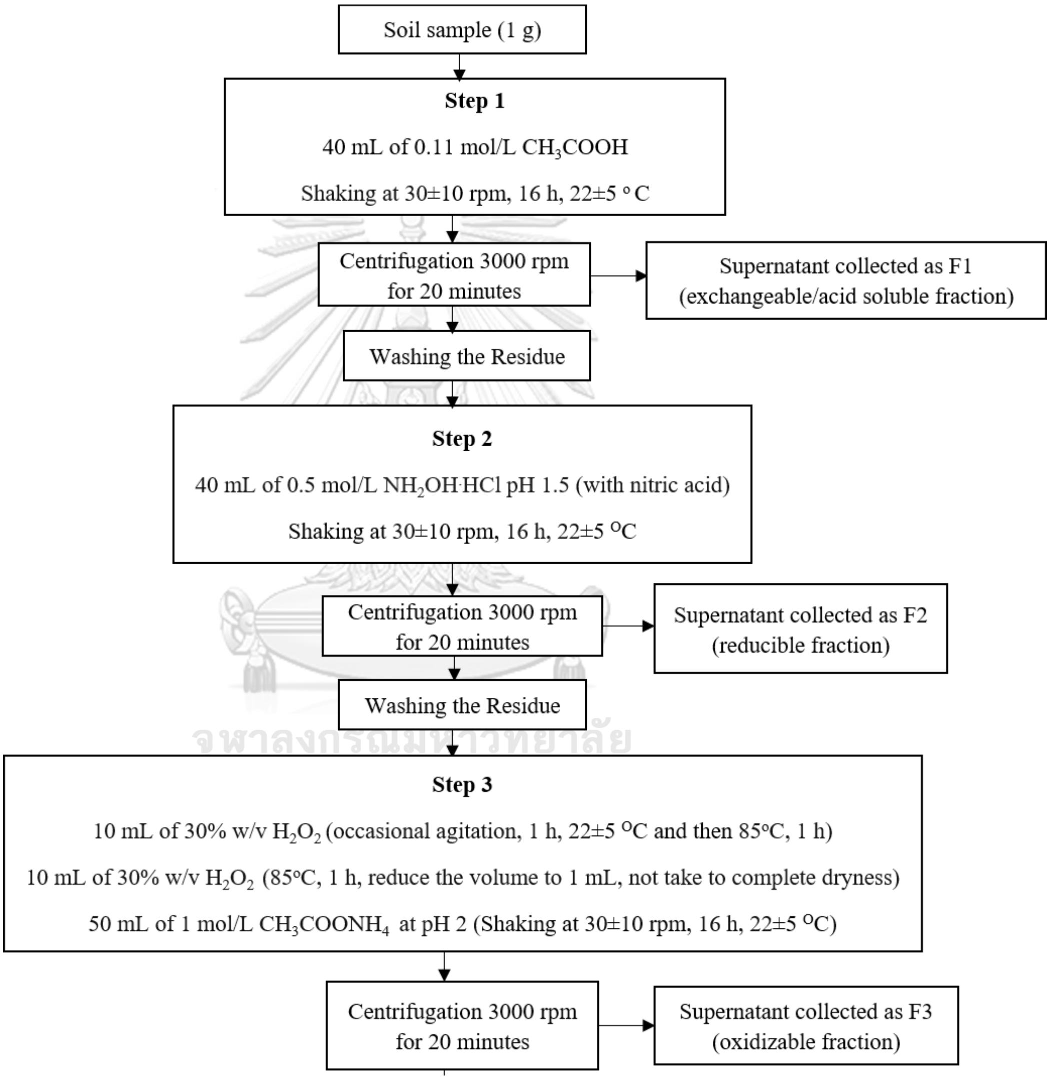


Figure 7 Modified BCR sequential extraction scheme

3.3.5 Quality assurance and quality control

To ensure analytical quality, the validity of the analyzing procedure was checked using the MESS-3, a certified marine sediment reference materials from National Research Council of Canada. MESS-3 was used for method validation by identifying the percentage of recovery which can be calculated by equation 3.2. For the percentage recovery, the percentage recoveries of all studied heavy metal were in the acceptable range of 80-120% (Addis & Abebaw, 2017). The percentage recovery of all heavy metals are shown in appendix B.

$$\text{Recovery (\%)} = \frac{A}{B} x\ 100 \qquad (3.2)$$

Where;

A is the heavy metal concentration measured from MESS-3

B is the reference heavy metal concentration of MESS-3

The limit of detection (LOD) for each heavy metal was determined from the analysis of 12 replicates of method blanks and was calculated by equation 3.3. In addition, the limit of quantification (LOQ) was obtained from a triplicate analysis of 12 method blanks and was calculated by equation 3.4. The LOD and LOQ of the heavy metals are shown in appendix B.

$$\text{LOD} = 3\text{x}S_{bl} \qquad (3.3)$$

$$\text{LOQ} = 10\text{x}S_{bl} \qquad (3.4)$$

Where; S_{bl} is the standard deviation of the method blank

To test the repeatability of the analysis of heavy metal in the study, five replicate readings of standard solutions were determined to identify relative standard deviation (RSD) that could be obtained as equation 3.5. The RSD of all studied elements was lower than 15% that suggested the precision of the study (appendix B).

$$\text{RSD (\%)} = \frac{standard\ deviation}{mean\ value}\ x\ 100 \qquad (3.5)$$

3.4 Risk Assessment

3.4.1 Potential ecological risk assessment

To quantitatively express the potential risk of the heavy metals in the soil, the potential ecological risk index (PERI) was applied in the study. This represents the sensitivity of the biological community to the heavy metal in the soil and shows the potential ecological risk affected by the overall contamination (Hakanson, 1980). The ecological risk factors (Er) of the individual heavy metals, and the PERI, which is a summation of Er, can be calculated following equations 3.6 and 3.7, respectively.

$$\text{Er} = \text{Tr x } (C_{sample}/C_{background}) \qquad (3.6)$$

$$\text{PERI} = \text{Er}_1 + \text{Er}_2 + \dots \text{Er}_n \qquad (3.7)$$

Where;

T_r = the biological toxic response factor of an individual element (As=10, Cd=30, Cu=Ni=Pb=5, Cr=2, and Mn=1, and Zn=1) (Hakanson, 1980)

C_{sample} = the total concentration of heavy metal in the soil sample (mg/kg)

$C_{background}$ = the total concentration of heavy metal in the control area (mg/kg)

The calculation of Er and PERI posed by the heavy metals can be categorized and summarized the risk levels as presented in Table 9.

Table 9 Potential ecological risk levels

Er	**PERI**	**Risk levels**
Er<40	PERI<150	Low potential ecological risk
40≤ Er<80	150≤ PERI<300	Moderate potential ecological risk
80≤ Er<160	300≤ PERI<600	Considerable potential ecological risk
160≤ Er<320	PERI>600	Very high potential ecological risk
Er>320	-	Extreme potential ecological risk

3.4.2 Risk assessment code

The risk assessment code (RAC) was used to determine the environmental risk of heavy metals, and RAC can be classified into five levels as displayed in Table 10 based on the percentage of the exchangeable and weak acid soluble fractions (equation 3.8) which are able to present the high potential bioavailability in the environment causing serious environmental concern (Perin et al., 1985). The levels of risk assessment code are shown in Table 10.

The percentage of exchangeable and weak acid soluble fraction = $\frac{A}{B}x100$ (3.8)

Where;

A = the concentration of heavy metal in exchangeable and weak acid fraction (F1) (mg/kg)

B = the total concentration of heavy metal (mg/kg)

Table 10 Classification of risk assessment code (RAC)

Criteria	**Risk**
<1	No risk
1-10	Low risk
11-30	Medium risk
31-50	High risk
>50	Very high risk

3.5 Data analysis

3.5.1 Statistical analysis

The difference of each total heavy metal concentrations and soil properties in different soil depths, including different seasons was analyzed by paired-T test. The mean of each total heavy metal concentration among different sites was compared the statistical differences using one-way ANOVA by Tukey HSD method. In addition, the correlation between heavy metals and heavy metals and soil properties was analyzed by Pearson's correlation. The level of significance was set at $p < 0.05$. All statistical analysis was conducted using SPSS (version 23) software package for Windows.

3.5.2 Enrichment factor

The enrichment factor (EF) was applied in the study in order to assess the contamination of metal in the soil as a result of human activities. The EF can be calculated by equation 3.9 :

$$EF= \frac{(C_x/C_{Mn})sample}{(C_x/C_{Mn})Reference} \qquad (3.9)$$

Where;

C_x = the concentration of metal x in soil samples (mg/kg)

C_{Mn} = the concentration of Mn in soil samples (mg/kg)

Overall, the equation of EF is the ratio of metal x content to the content of Mn in the soil sample and the reference, respectively. In this study, Mn was selected as a reference metal because Mn was the most abundant element in the soils in the study area, and Mn showed similar ranges of the concentration in all sampling sites. Moreover, Mn was considered as the main crust-derived element (Wedepohl, 1995). With the same reasons, many previous studies selected Mn as a reference element in the enrichment factor calculation (Fujimori et al., 2012; Xu et al., 2015; W. Zhao et al., 2015). After the calculation of EF was done, the categories of EF values of the metals is listed in Table 11.

Table 11 Categories of enrichment factor

EF values	Categories
<2	Minimal enrichment
2-5	Moderate enrichment
5-20	Significant enrichment
20-40	Very high enrichment
>40	Extremely high enrichment

3.5.3 Principal component analysis

PCA was used to identify the possible sources of the heavy metals in the soils in the study area. PCA, one of the multivariate statistical techniques, was applied to identify and differentiate the possibility of heavy metals studied in the study area. The total heavy metal concentration was subjected to the PCA aiming to get groups or principal components containing associated heavy metals. PCA was analyzed by means of SPSS (version 23) software package for Windows.

3.5.4 Geostatistical analysis

In order to identify the hot-spot of heavy metal pollution in soils, GIS-based spatial distribution mapping was applied to determine possible hot-spots and spatial distribution patterns of combined metal pollution risk posed by the individual heavy metals in the e-waste dismantling and burning sites and the surrounding areas. Spatial distribution maps of the ecological risk factor (Er) in the study area were created by ArcMap 10.7.1 software with the method of Inverse Distance Weighted (IDW) technique.

Chapter 4

Results and discussions

Chapter 4 is composed of the results of heavy metal concentration in soils, heavy metal fractionation in soils, and risk assessment posed by all the studied heavy metals. The results of these are shown in 4.1, 4.2, and 4.3, respectively. In addition, apart from the total heavy metal concentrations, 4.1 contained soil properties in the study area, principal component analysis (PCA), heavy metal contribution in the soils, and enrichment factors of heavy metals. 4.2 showed the results of heavy metal fractionation in soils, and 4.3 is the results of risk assessment caused by the heavy metals in the study area.

4.1 Heavy metal concentration in soil in the e-waste dismantling site of Buriram, Thailand

4.1.1 Soil properties in the study area

Soil properties in the study area were presented in Table 12. According to soil pH, pH ranged from 4.94-5.01 in the reference (ref) and 5.01-7.03 in the control sites. For non-e-waste dismantling houses (NE), pH ranged between 4.44 and 7.80. In groundwater-well area (GW), pH ranged from 4.97 to 7.97. In e-waste dismantling houses (EW), pH was in the range of 4.38 to 7.87. For paddy fields (PF), pH ranged between 4.09 and 8.71. In open-burning sites (OB), pH ranged from 7.01 to 7.90. pH values measured in the sampling sites revealed that the pH values were varied from strong acid to mild alkaline in dry season and wet season; however, soil pH in this study was site-dependent.

According to Alloway (2012) and Rieuwerts et al. (1998), metal solubility in soils tends to increase at low soil pH, while metal solubility in soils decrease at higher soil pH level. Soil pH has an association with metal adsorption in soils in term of competition of H^+ ions. Adsorption sites with low pH cause decreasing of metal adsorption. Moreover, acid condition in soils leads to the dissolution of Fe-Mn oxides, and their adsorption sites. For this reason, the adsorption of metals on Fe-Mn oxides could decrease, so the acid condition in soils leads to the solubility of metal in soils.

Considering oxidation-reduction potential (ORP) in the study area, all ORP values measured from all soil samples were in the range between 120 and 414 mV. With this range of ORP, the soils in the study area was defined as moderately reduced soil (Essington, 2015; Fiedler et al., 2007; Scholz, 2019). Under the moderately reducing or suboxic condition, oxygen followed by NO_3^-, Mn^{4+} and Fe^{3+} are the order of the preferred electron acceptors in redox process in soils (Fiedler et al., 2007). Higher ORP is generally recorded in well aerated soils. In contrast, ORP in soils in waterlogged soil and organic matter-enriched soil tends to be low (Rieuwerts et al., 1998). Metals are more readily dissolved in soil with lower ORP (waterlogged soils) because under reducing conditions, the dissolution of Fe-Mn oxyhydroxides results in the release of adsorbed metals on the Fe-Mn oxides occurring as coating on clay minerals (Chuan et al., 1996).

Soil organic matter contents (OM) were varied among the sampling sites; OM in the Ref ranged between 3.56 and 4.48%, while OM in the control site ranged from 3.75 to 4.81%. For the OM in NE, OM had a range between 3.35 and 6.58%. Additionally, OM ranged from 4.25 to 7.13% in GW. For EW, OM ranged between 3.35 and 6.58%, and OM had a wide range between 2.85 and 6.35 in the PF. Lastly, OM in the OB showed OM ranging between 3.94 and 5.83%. OM typically accumulates at the soil surfaces, and OM in soils has a significant influence on metal binding and metal retention through complexation and adsorption processes. Metals tend to bind with organic components contained in OM in the soils and solution phase of soil, and subsequently become organo-metal complexes (Alloway, 2012; Rieuwerts et al., 1998; Wang et al., 2015).

Lastly, all soil samples contained a sand fraction of more than 50%, which means that the soils in the study area were sand-enriched soils. The soil types found in the study area were sand, sandy loam, loamy sand, and sandy clay loam. Different soil fraction showed different affinity of metal adsorption; the strong affinity of metal adsorption to soil fraction is the decreasing order as follows: clay> silt > sand (Andersson, 1979). For this reason, available metals were recorded in the sand fraction rather than clay fraction (Qian et al., 1996); the accumulation of metals in the clay fraction is attributed to the high surface area for metal adsorption and also high presence of clay minerals, organic matters, Fe-Mn oxides, and sulphides (Rieuwerts et al., 1998).

Table 12 Soil properties in the study area

Soil samples	**layers**		**Soil properties**						
			pH		**ORP (mV)**		**OM (%)**		**Soil type**
			Dry	Wet	Dry	Wet	Dry	Wet	
Ref	surface	Mean±SD	NA	4.96±0.02	NA	366±19.48	NA	4.47±0.021	Sandy loam
		Range	NA	4.94-4.98	NA	354.20-389.30	NA	4.45-4.48	
	Subsurface	Mean±SD	NA	5.00±0.01	NA	418±0.87	NA	3.57±0.014	Loamy sand
		Range	NA	4.99-5.01	NA	417.4-419	NA	3.56-3.58	
Control	surface	Mean±SD	5.40±0.21	6.74±0.27	272.13±5.93	382.96±4.98	4.33±0.42	NA	Loamy sand
		Range	5.30-5.68	6.50-7.03	265.6-277.2	377.3-386.7	4.01-4.81	NA	
	Subsurface	Mean±SD	5.03±0.03	6.05±0.01	301.43±20.71	252.96±0.05	3.91±0.276	NA	Loamy sand
		Range	5.01-5.07	6.05-6.06	285.3-324.8	252.9-253	3.75-4.23	NA	
NE	surface	Mean±SD	4.82±1.56	5.21±1.17	294.45±71.02	310.95±74.28	5.63±0.76	NA	Sandy loam
		Range	4.50-7.48	4.91-7.07	230.3-365.1	234.9-379.1	4.68-6.58	NA	
	Subsurface	Mean±SD	4.77±1.59	4.94±1.72	298±92.1	283.43±66.14	4.10±0.50	NA	Sandy loam
		Range	4.44-7.38	4.64-7.80	216.1-383.4	204.4-360.4	3.35-4.74	NA	
GW	surface	Mean±SD	6.50±0.71	6.73±0.26	265.28±31.23	321.58±43.12	5.68±0.62	NA	Loamy sand, sandy loam

Soil samples	layers		Soil properties						
			pH		ORP (mV)		OM (%)		Soil type
		Range	5.97-7.70	6.36-7.12	235.1-309.7	264.1-370.5	4.90-7.13	NA	
	Subsurface	Mean± SD	5.6± 1.09	6.29± 0.86	264± 32.50	281.09± 104.55	4.98± 0.49	NA	Loamy sand
		Range	4.97-7.56	5.72-7.97	228.8-304.7	156.9-392.1	4.25-5.99	NA	
EW	surface	Mean± SD	6.70± 0.19	5.78± 1.22	238.67± 53.02	283.33± 14.79	5.63± 0.76	NA	Loamy sand
		Range	6.50-6.95	5.48-7.87	190.4-286.3	269.6-297.1	4.68-6.58	NA	
	Subsurface	Mean± SD	4.71± 1.61	4.56± 0.17	282.27± 77.56	315.71± 51.99	4.10± 0.50	NA	Loamy sand, sand
		Range	4.38-7.40	4.42-4.75	215.9-350.8	262.9-365.6	3.35-4.74	NA	
PF	surface	Mean± SD	4.89± 1.00	5.32± 1.03	310.67± 44.07	294.17± 58.18	4.86± 0.88	NA	Sandy loam, loamy sand
		Range	4.46-7.67	4.66-7.41	254.9-372.6	198-386.6	3.32-6.35	NA	
	Subsurface	Mean± SD	5.23± 1.36	4.74± 1.29	275.63± 94.67	290.38± 62.56	4.76± 0.91	NA	Sandy loam, sandy clay loam
		Range	4.52-8.71	4.09-7.72	245.9-377.8	219-380.2	2.85-6.07	NA	
OB	surface	Mean± SD	7.32± 0.24	7.47± 0.33	207.12± 12.78	310.61± 33.57	4.72± 0.512	NA	Sandy loam
		Range	7.01-7.67	7.23-7.90	195.9-221.7	263.9-352.3	4.06-5.38	NA	
	Subsurface	Mean± SD	7.42± 0.27	7.41± 0.11	198.4± 15.94	236± 4.46	5.11± 0.721	NA	Sandy loam
		Range	7.20-7.76	7.24-7.52	184.1-212.6	233-244	3.94-5.83	NA	

"NA" = not available data

4.1.2 Total heavy metal concentrations in the study area

The concentrations of metals (i.e., As, Cd, Cr, Cu, Mn, Ni, Pb, and Zn) in soils taken from the reference site, control site, non-e-waste houses, groundwater-well area, e-waste dismantling household areas, paddy fields, and open-burning sites from Daeng Yai and Ban Pao village in both wet and dry season are presented as figures shown in sections 4.1.2.1-4.1.2.8. The results of statistical analysis, including ANOVA, and T-test are also explained in section 4.1.2.1-4.1.2.8 to investigate the significant difference of heavy metal concentration found in the soils of different land uses and the significant difference of heavy metal accumulation between in surface and subsurface soils. Intervention values being mentioned in this section are the representative of the level of contamination above which there is a severe case of soil contamination. The reference values of each heavy metals were the concentration of the metals in soils collected from the reference site. Additionally, the concentration of the heavy metals and the statistical analysis are shown in Appendix C.

4.1.2.1 As concentration in soils

As contents in the soil samples are presented separately by surface and subsurface layers, as shown in Figure 8 - Figure 9.

A. As in surface layer

Figure 8 shows the concentration of As in surface soil collected from six sampling sites and a reference site (5 km away from the e-waste sites). The wide range of 11.04 to 11.41 mg/kg at the control site, 4.79 to 10.22 mg/kg for non-e-waste sites, 4.48 to 12.62 mg/kg for groundwater-well area, 3.23 to 10.23 for e-waste dismantling household area, 2.01 to 29.50 mg/kg for paddy fields, 10.58 to 18.82 mg/kg for open-burning sites, were found, and the concentration at the reference area was 9.73 mg/kg. As concentration at the reference site was in the average world range of As in uncontaminated soils, ranging from 0.1 to 55 mg/kg (Alloway, 2012). Compared to the reference As value (9.73 mg/kg), As in all the samples taken from the control site and the open-burning site exceeded the reference value. However, As concentration in the study area did not exceed the intervention value set at 55 mg/kg, but the As in all sampling sites exceeded the Thai standard for residential and agricultural soils set at 3.9 mg/kg. Considering the distribution of the As contents in the surface soil, the highest As were found at the open-burning site (16.41 mg/kg), closely followed by those observed at the paddy field (11.95 mg/kg), non-e-waste sites (8.30 mg/kg), groundwater well (7.89 mg/kg), and e-waste sites (6.76 mg/kg). According to the result of statistical analysis using one-way ANOVA, there was no significant difference between the As concentration among all sites ($p>0.05$).

B. As in subsurface layer

According to Figure 9, As contents in the subsurface layer had wide ranges in this study. As at the control site was in the range of 12.75 – 15.55 mg/kg, 4.99 – 11.43 mg/kg at the non-e-waste site, 4.75 – 8.48 mg/kg at the groundwater-well area, 1.62-9.57 mg/kg at the e-waste site, 1.44 – 39.08 mg/kg at paddy field, and 10.68-27.59 at the open-burning site. The concentration of As at the reference area was 7.56 mg/kg. Like the As reference value in surface soil, As reference value in subsurface soil was also in the average world range of As in uncontaminated soils (0.1-55 mg/kg). Based on the As reference value (7.56 mg/kg), As concentrations in all soil samples taken from the control site and open-burning site were greater than that in the reference area, which was the same as in the surface soil.

Comparing the As in subsurface soil with the intervention value regulated by Netherlands Ministry of Housing, Spatial Planning and the Environment, As in the soil from all the sites did not over the intervention value (55 mg/kg), but As in all sampling sites was found to exceed the Thai standard for residential and agricultural soils set at 3.9 mg/kg. As shown in Figure 4.2, the obvious trend was observed among the sites; the concentration of As shown in the following direction: open-burning (18.91 mg/kg) > paddy field (18.02 mg/kg) > non-e-waste (9.07 mg/kg) > groundwater-well area (5.92 mg/kg) > e-waste site (5.77).

Regarding the statistical analysis, the As concentrations in subsurface soil from the groundwater-well area and paddy field were significantly different ($p<0.05$); subsurface As concentrations in paddy fields were significantly higher than those in the groundwater-well area. Considering the As concentrations difference with soil layers using paired T-test analysis, there was a significant difference observed at paddy fields between the As contents in surface and subsurface. The high concentration of As in the paddy fields might be attributed to the fertilizer and insecticide application during the rice planting; according to Alloway (2012), inorganic As was found to be in pesticides, so long term application of the inorganic arsenical pesticide might cause As accumulation in the paddy soil.

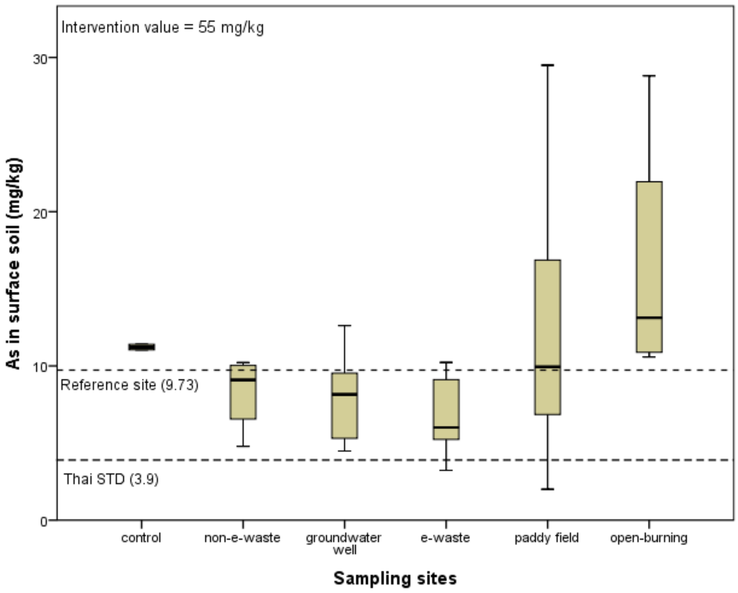


Figure 8 As concentrations in surface soils collected from the study area

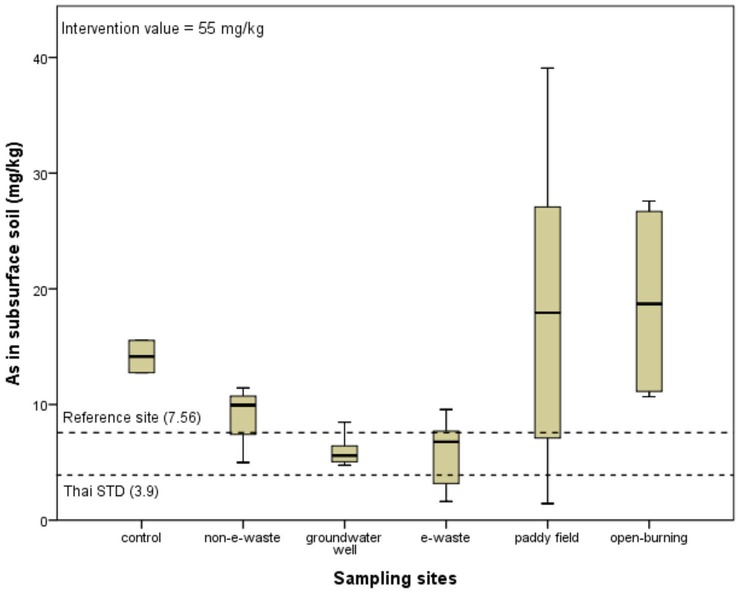


Figure 9 As concentrations in subsurface soils collected from the study area

4.1.2.2 Cd concentration in soils

A. Cd in surface layer

As shown in Figure 10, Cd concentrations in surface soils were varied with the different sampling sites. Cd in soils ranged 0.62 to 0.65 mg/kg at control site, 0.26 to 0.46 mg/kg at non-e-waste site, 0.12 to 0.53 mg/kg at groundwater-well area, 0.16 to 0.65 mg/kg at e-waste dismantling site, 0.11 to 1.70 mg/kg at paddy fields, and 0.51 to 2.29 mg/kg at open-burning site. At the reference site, Cd concentration was

0.45 mg/kg. The reference Cd concentration of the study area (0.45 mg/kg) was in the range of Cd measured in non-contaminated soils that are typically found between 0.1 – 1.0 mg/kg (Alloway, 2012). After comparing Cd contents of the soil samples with the Cd reference value, Cd in all samples from the control site and open-burning site was greater than the reference value. Nevertheless, Cd concentrations in all samples in this study did not exceed both the intervention value (12 mg/kg) and Thai standard for residential and agricultural soils (37 mg/kg) (Thai Office of National Environment Board, 2014).

The trend of Cd concentrations in surface soils was observed in the following decreasing order: open-burning site (1.24 mg/kg) > paddy fields (0.68 mg/kg) > e-waste dismantling house (0.40 mg/kg) > groundwater well area (0.37 mg/kg) > non-e-waste house (0.35 mg/kg). Cd concentration at the open-burning site was statistically significant increase than that of groundwater-well, non-e-waste, and e-waste dismantling sites ($p<0.05$).

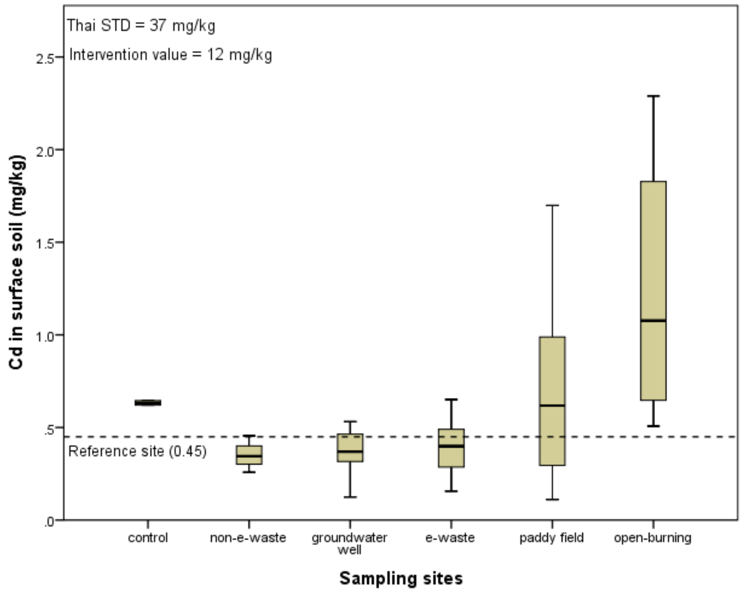


Figure 10 Cd concentrations in surface soils collected from the study area

B. Cd in subsurface layer

Cd concentrations measured in subsurface soils were shown in Figure 11**.** Cd contents ranged from 0.69 to 0.76 mg/kg at control site, 0.25 to 0.49 mg/kg at non-e-waste site, 0.02 to 0.46 mg/kg at groundwater well area, 0.11 to 0.49 mg/kg at e-waste site, 0.09 to 1.73 mg/kg at paddy fields, 0.51 to 10.93 at open-burning site. Cd with a concentration of 0.35 mg/kg was found at the reference area, which was also in the typical range of Cd in uncontaminated soil (0.1-1.0 mg/kg) (Alloway, 2012). The results showed that a few samples in non-e-waste, groundwater well, e-waste, and paddy fields had elevated Cd contents compared to the Cd in the reference area, whereas Cd concentrations in all samples collected from control site and open-burning site were above reference Cd contents of the study area. There was no Cd in any samples exceeded the Thai standard and intervention values.

With respect to the distribution trend of Cd found in the subsurface soils, Cd was found highest at open-burning site (3.46 mg/kg) followed by paddy-fields (0.89 mg/kg), non-e-waste site (0.42 mg/kg), groundwater well (0.30 mg/kg), and e-waste dismantling site (0.29 mg/kg), respectively. The statistical analysis revealed that there was no significant difference in Cd in subsurface soils among the sites. On the other hand, the analysis of the Cd difference between in surface and subsurface showed that surface-soil Cd at e-waste dismantling house was significantly higher than subsurface-soil Cd ($p<0.05$).

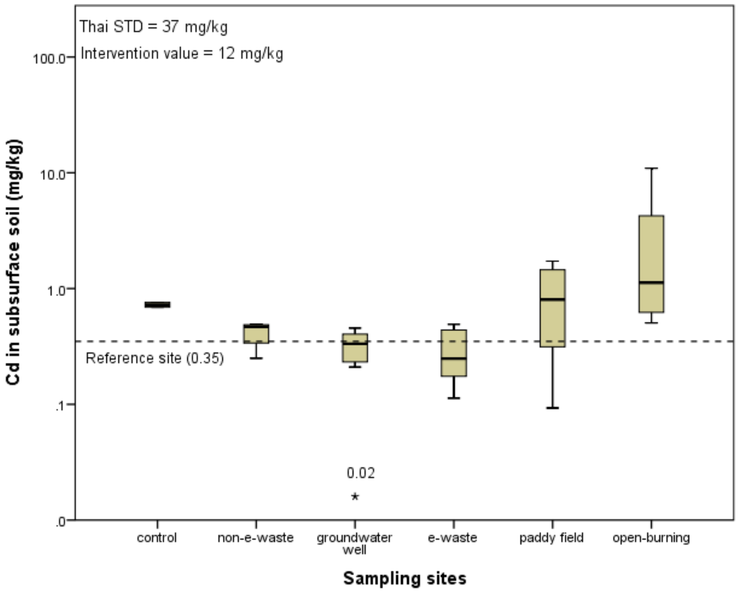


Figure 11 Cd concentrations in subsurface soils collected from the study area

4.1.2.3 Cr concentration in soils

A. Cr in surface soil

Cr concentrations in surface soil are depicted in Figure 12. Cr had wide ranges with different sites. Cr ranged 18.49 to 29.12 mg/kg for control site, 3.75 to 19.22 mg/kg for non-e-waste site, 1.03 to 20.71 mg/kg for groundwater well area, 3.68 to 24.09 mg/kg for e-waste site, 0.37 to 49.49 mg/kg for paddy fields, and 15.35 to 65.84 for open-burning site. For the reference area, Cr was 11.61 mg/kg. The Cr reference concentration was in the range of the Cr background concentrations of soil on the basis of the parent material studied by Alloway (2012) (5-68 mg/kg). Comparing Cr contents in the sampling sites to the reference value, the result turned out the same as the results of As and Cd, where Cr in all soil samples taken from the control area and open-burning site was over the Cr reference value. Cr concentrations in all sampled soils met the Thai standard for residential and agricultural soils (300 mg/kg) and intervention value (380 mg/kg). The tendency of Cr concentration in the study site was observed in this decreasing order: open-burning site (31.15 mg/kg) > paddy fields (19.70 mg/kg) > e-waste site (12.56 mg/kg) > non-e-waste site (11.94 mg/kg) > groundwater well area (9.46 mg/kg). The result of the ANOVA test showed no significant difference in Cr among the sampling sites

B. Cr in subsurface soil

Cr concentration measured from subsurface soils in the study site was presented in Figure 12. The ranges of Cr contents were broad, depending on the sampling sites. Cr had a range from 20.93 to 33.54 mg/kg in control site, 3.62 to 25.93 mg/kg in non-e-waste site, 1.39 to 13.28 mg/kg in groundwater well area, 3.57 to 24.37 mg/kg in e-waste area, 0.65 to 44.05 mg/kg in paddy fields, and 16.02 to 56.46 mg/kg in open-burning site. The reference value of Cr in the subsurface was 9.99 mg/kg. Similar to the results of Cr in surface soil, Cr in all samples taken from the control site and open-burning site was higher than reference Cr in the study area. Comparing the Cr contents to the Thailand standard (300 mg/kg) and intervention value (380 mg/kg), Cr in all samples did not exceed the standards.

The pattern of Cr concentrations in subsurface soils was observed in the descending order: open-burning (34.89 mg/kg) > paddy fields (22.61 mg/kg) > non-e-waste site (15.18 mg/kg) > e-waste site (11.23 mg/kg) > groundwater-well area (7.60 mg/kg). Regarding the statistical analysis, Cr in subsurface layer at the open-burning site was significantly higher than groundwater well area ($p<0.05$), while the significant difference between Cr in surface and subsurface layer at any sampling sites was not found.

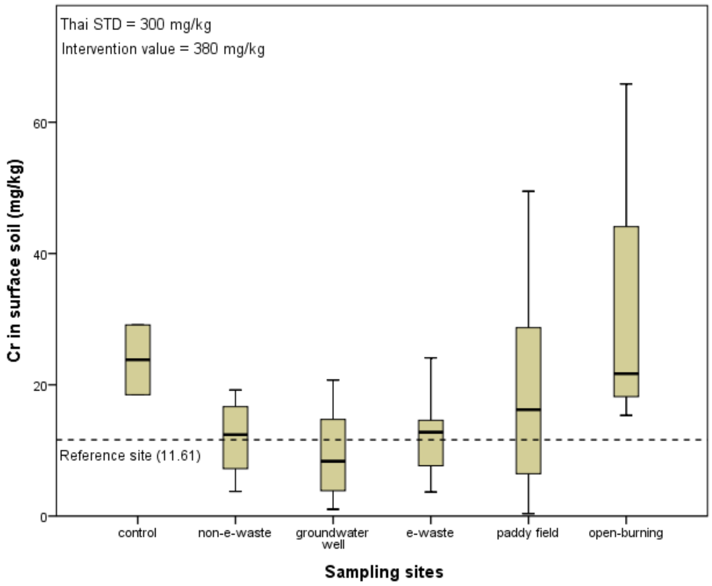


Figure 12 Cr concentrations in surface soils collected from the study area

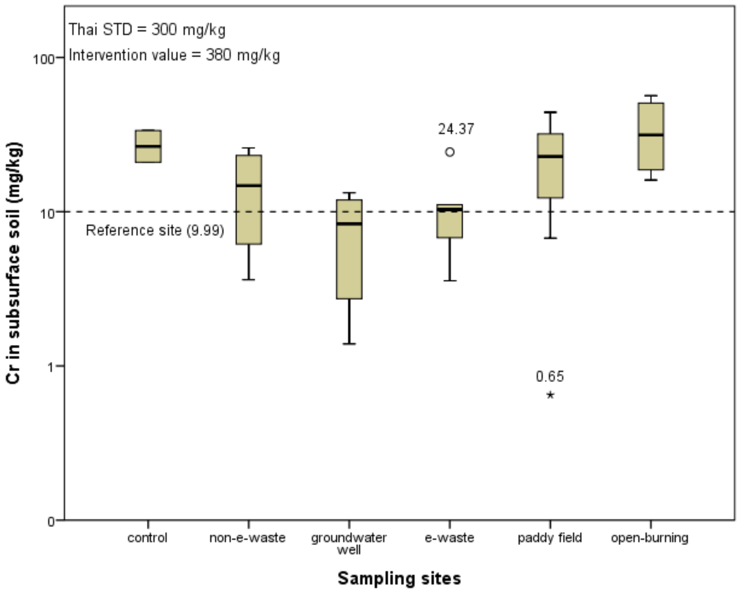


Figure 13 Cr concentrations in subsurface soils collected from the study area

4.1.2.4 Cu concentration in soils

A. Cu in surface soil

According to Figure 14, Cu concentrations in surface soils ranged widely in the study site; Cu ranged from 6.42 to 9.38 mg/kg at control site, 0.34 to 36.85 mg/kg at non-e-waste site, 0.14 to 11.97 mg/kg at groundwater-well area, 3.34 to 60.99 mg/kg at e-waste dismantling site, 0.34 to 91.13 mg/kg at paddy field, and 54.17 to 1,725.45 mg/kg at open-burning site. For the reference site of the study area, Cu was 6.02 mg/kg. The reference Cu was in the range of background Cu concentration, ranging between 2 and 50 mg/kg, reported by Alloway (2012). In addition, the reference Cu value was similar to the Cu background value for Thailand (7.60 mg/kg) reported by Zarcinas et al. (2004). Cu in all samples from the control site and open-burning site was higher than the reference value by approximately 1.5 times and 9 - 287 times, respectively.

Cu in almost all soil samples collected from the open-burning site exceeded the intervention value set as 190 mg/kg. The results indicated that Cu contamination in surface soil had become a problem in the open-burning site; as a result, environmental remediation should be advised to the open-burning area. Cu in surface soil was tended to decrease by descendent order: open-burning site (753.84 mg/kg) > e-waste dismantling site (28.23 mg/kg) > paddy-field (19.51 mg/kg) > non-e-waste (10.20 mg/kg) > groundwater well area (4.09 mg/kg). The statistical analysis using ANOVA showed that Cu in surface soil from the open-burning site was significantly higher than that in the other sampling sites ($p<0.05$). The present of high Cu concentrations at OB obviously implied to severe impact on risk assessment.

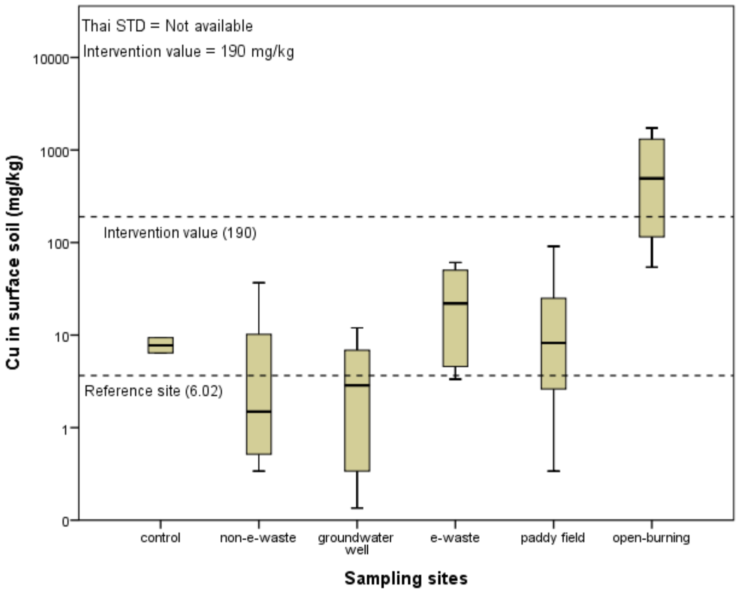


Figure 14 Cu concentrations in surface soils collected from the study area

B. Cu in subsurface soil

Cu in subsurface soil are depicted in Figure 12. Cu ranged 6.28 to 9.86 mg/kg at control site, 0.34 to 5.88 mg/kg at non-e-waste site, 0.14 to 5.11 mg/kg at groundwater-well area, 0.34 to 7.73 mg/kg at e-waste dismantling site, 0.34 to 57.42 mg/kg at paddy fields, 10.30 to 24,431.56 mg/kg at the open-burning sites. For the soil at the reference site, Cu was 3.98 mg/kg, which was within the board range of background Cu concentration, ranging between 2 and 50 mg/kg, reported by Alloway (2012). Comparing Cu concentration in all samples to the Cu reference value revealed that Cu in all samples from the control site and open-burning site was higher than the Cu reference value, whereas Cu contents in most samples taken from the rest of the sampling sites were lower than the reference Cu. Regarding the Cu concentration in subsurface soil from the open-burning site, the minimum and maximum concentration of Cu was 2.5 to 6,140 times higher than the reference value.

According to the intervention value of Cu (190 mg/kg), a soil sample from the open-burning site, which had 24,431.56 mg/kg of Cu, was considered as a seriously contaminated site, and need to be remediated as Cu contamination could possibly threaten humans, plants and animals in the open-burning area (VROM, 2000). The particular trend of Cu contents in subsurface soil was observed among the sampling sites in the following order: open-burning site (6,157.70 mg/kg) > paddy-fields (13.10 mg/kg) > non-e-waste site (2.81 mg/kg) > groundwater well area (1.27 mg/kg). However, a significant difference in Cu with different sampling sites was not found as well as the difference between surface and subsurface.

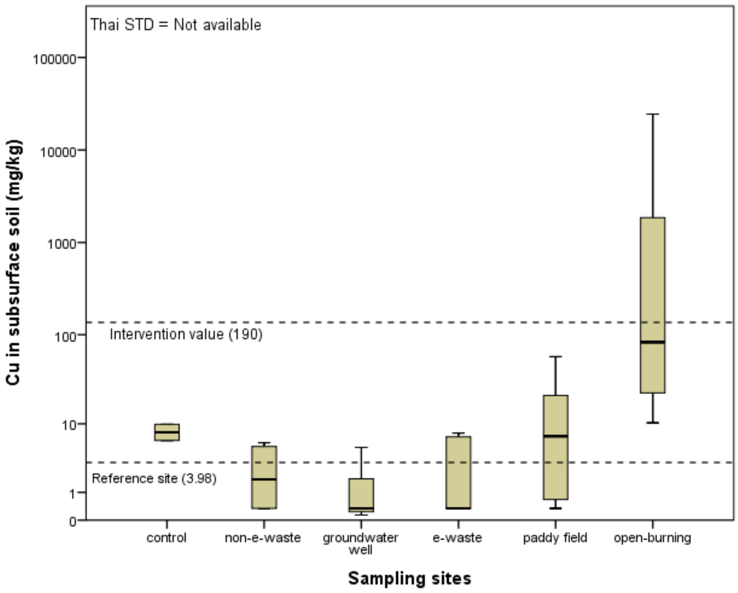


Figure 15 Cu concentrations in subsurface soils collected from the study area

According to the Cu contamination in both surface and subsurface soil at the open-burning site, the extremely high Cu concentration in soil was a result of an informal open burning of coated Cu cables. Based on the field observation, workers typically burn the wire cables directly on bare ground at the burning site in order to get the copper for sale. Open-burning of Cu cables generates Cu-rich particulates, and then the particulates can be deposited onto the surface soil. In the meantime, the fine fragments from the burning activity also remain at the site (Ackah, 2017). Therefore, the uncontrolled burning of Cu wires is a potential activity resulting in Cu contamination in the open-burning site.

4.1.2.5 Mn concentration in soils

A. Mn in surface soil

As shown in Figure 16, Mn had wide ranges within the sampling sites. Mn ranged 41.52 to 66.03 mg/kg in control site, 19.07 to 121.18 mg/kg in non-e-waste site, 22.92 to 175.74 mg/kg in groundwater-well area, 46.76 to 168.10 mg/kg in e-waste dismantling site, 0.82 to 296.29 mg/kg in paddy field, and 47.55 to 365.83 in open burning site. For the reference area, Mn was 304.14 mg/kg, which was sufficiently high compared to Mn in the other sampling sites. However, the Mn reference value was slightly low compared with Alloway (2012) and Wedepohl (1995) studies, that reported 850 mg/kg of Mn were geochemically found in soils, and 527 mg/kg of Mn were naturally found in the upper continental crust which is approximately 21 km in depth, respectively. The result of comparing the Mn concentration in all samples to the Mn reference value showed that only one sample collected from the open-burning site had an exceeded Mn. Based on Thailand guidance value of Mn (1,800 mg/kg), Mn concentration in soil samples met Thailand guidance. The particular trend of Mn in surface soil was observed in this descending order: open-burning site (183.25 mg/kg) > paddy-field (104.42 mg/kg) > e-waste dismantling site (103.92 mg/kg) > groundwater-well area (97.83 mg/kg) > non-e-waste site (64.95 mg/kg). However, there was no difference in Mn concentration within the different sampling sites.

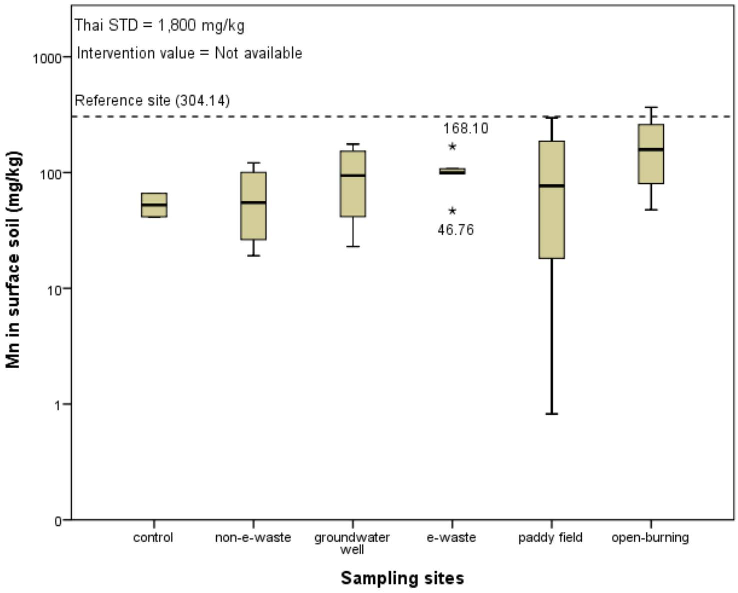


Figure 16 Mn concentrations in surface soils collected from the study area

B. Mn in subsurface soil

Figure 17 shows Mn concentration in subsurface soils in the study area; Mn had a wide range in the subsurface soils. Mn was 30.77 to 41.10 mg/kg at control site, 19.16 to 121.89 mg/kg at non-e-waste site, 7.57 to 338.52 mg/kg at groundwater well are, 14.38 to 125.97 mg/kg at e-waste dismantling site, 0.16 to 548.05 mg/kg at paddy field, and 86.50 to 338.13 mg/kg at open-burning site. For the reference site, Mn was 108.35 mg/kg at subsurface soil. Similar to the result of reference Mn in surface soil, Mn in subsurface soil was lower than the Mn contained in the upper continental crust. Mn in all soil samples compared to that of reference soil, non-e-waste site, groundwater well area, e-waste dismantling site showed barely exceed than the reference soil, while Mn in more than half of the samples taken from paddy field and open-burning site was above the reference Mn level.

Regarding the range of Mn contents in surface and subsurface soil at the paddy fields, the wide range of Mn contents are mainly influenced by soil parent material variations, and agricultural activities variation (Kong et al., 2018; K. Zhao et al., 2015). According to ThaiDMR (2010), sandstones and shales are the main types of soil parent materials reported in the study area. Sandstones, which are coarse-grained sediments, contain relative low Mn (100 mg/kg) , while shales, fine-grained materials, contain high concentrations of Mn (850 mg/kg) (Alloway, 2012). During soil development processes, weathering of various types of the parent materials results in the heterogeneity of soil Mn content in the paddy soils. Apart from the parent materials variation, agricultural activities are a crucial factor influencing the Mn accumulation in paddy soils; Mn as a micronutrient is composed in phosphatic, and lime fertilizers in the range of 40-2,000 mg/kg, and 40-1,200 mg/kg, respectively. As the contents of Mn in the fertilizers are relatively high, regularly fertilized paddy soil enhances the accumulation of great amounts of Mn in the paddy soils (Alloway, 2012; Kong et al., 2018). However, the fertilizer application at the paddy fields was not monitored in this study.

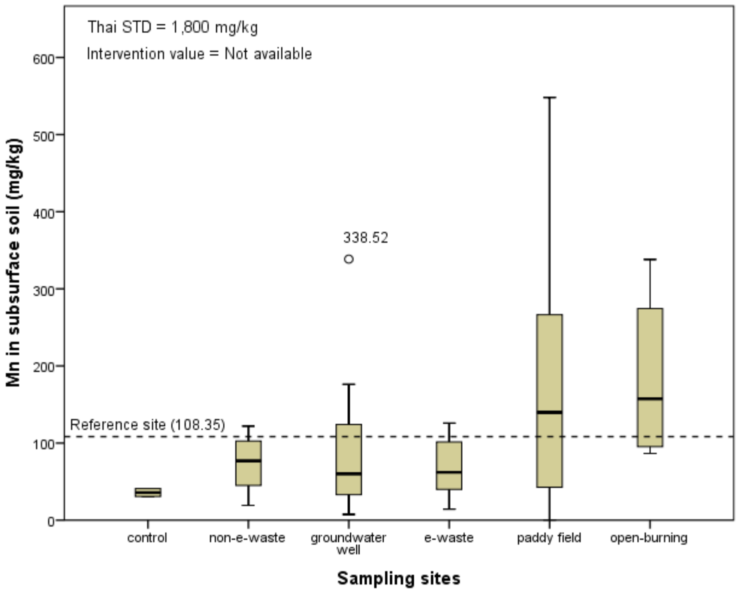


Figure 17 Mn concentrations in subsurface soils collected from the study area

According to the soil guidance of Thailand set at 1,800 mg/kg, Mn in all samples did not exceed the guidance value. The particular trend found in the subsurface soils was slightly different from the trend in the surface soil. The concentration of Mn in the subsurface soil was in the following order: open-burning site (184.87 mg/kg) > paddy fields (171.75 mg/kg) > groundwater well area (97.63 mg/kg) > non-e-waste site (73.75 mg/kg) > e-waste dismantling site (68.75 mg/kg). However, there is no significant difference in Mn found among different sampling sites. For the difference between Mn in the surface and subsurface soils, at e-waste dismantling houses, Mn in the surface soils was significantly higher than in the subsurface soils ($p<0.05$).

4.1.2.6 Ni concentration in soils

A. Ni in surface soil

Ni concentration in surface soil collected from the sampling site is shown in Figure 18. Ni was varied among the sampling sites; Ni ranged from 5.85 to 6.53 mg/kg at the control site, 3.10 to 4.20 mg/kg at the non-e-waste site, 4.08 to 6.81 mg/kg at groundwater well area, 2.33 to 8.12 mg/kg at e-waste dismantling site, 1.22 to 14.82 mg/kg at paddy field, and 5.17 to 17.89 mg/kg at the open-burning site. At the reference site, Ni was 7.09 mg/kg in surface soil. According to Alloway (2012), Ni content presents a broad range in soils, mainly from 0.2 to 450 mg/kg, with the world mean of 22 mg/kg. Besides, Wedepohl (1995) reported that approximately 18.6 mg/kg of Ni generally presents in the upper continental crust, and Zarcinas et al. (2004) also reported 14.7 mg/kg of Ni was the background value of Thailand soils. However, the Ni reference value in this study (7.09 mg/kg) was lower than those in other studies.

Comparing Ni in surface soil samples with the Ni reference value, 20% of soil samples from the e-waste dismantling site, and each 50% of soil samples from the paddy field and open-burning had exceeded Ni in the surface soil. Based on Thai soil standard of Ni (1,600 mg/kg), and the intervention value (210 mg/kg), Ni in all soil samples met both standards. As shown in the Figure 18, a particular trend of Ni in surface soil among the sampling sites was observed in the following descending order: open-burning (10.03 mg/kg) > paddy field (7.08 mg/kg) > e-waste dismantling site (5.56 mg/kg) > groundwater well area (4.98 mg/kg) > non-e-waste site (3.73 mg/kg). Nonetheless, a significant difference of Ni in surface soil within the different sampling was not found.

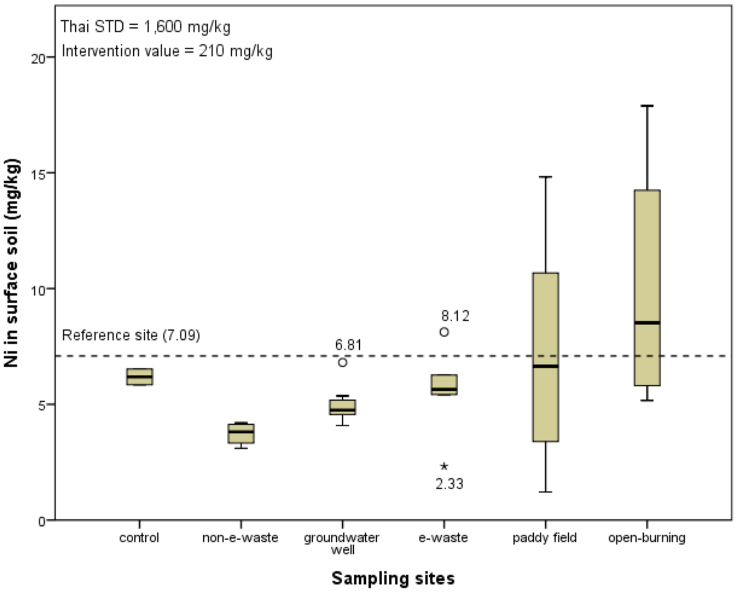


Figure 18 Ni concentrations in surface soils collected from the study area

B. Ni in subsurface soil

Ni concentration in subsurface soil taken from the sampling sites was presented in Figure 19. There was a wide range of Ni in the study area. Ni ranged 8.07 to 8.10 mg/kg at control site, 3.28 to 5.98 mg/kg at non-e-waste site, 2.64 to 5.19 mg/kg at groundwater-well area, 1.77 to 4.34 mg/kg at e-waste dismantling site, 1.13 to 23.19 mg/kg at paddy fields, and 5.02 to 94.17 mg/kg at open-burning. For the reference site, Ni was 5.50 mg/kg, which was lower than the studies of Alloway (2012), Wedepohl (1995), and Zarcinas et al. (2004). The comparison between subsurface Ni in the reference site and the other sampling site revealed that all samples from the control site and some samples from the non-e-waste site, paddy fields, and open-burning had Ni concentrations that exceeded the reference Ni value. However, Ni concentrations in all soil samples were not above the soil standard of Thailand and the intervention value.

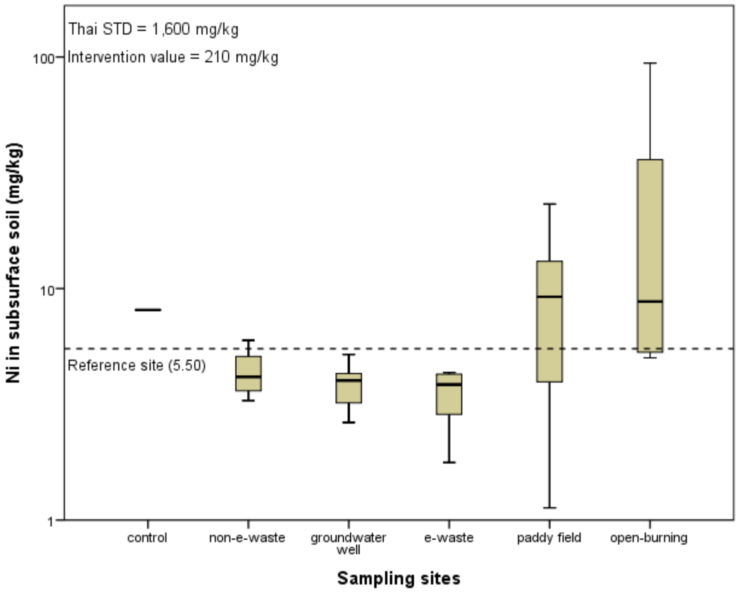


Figure 19 Ni concentrations in subsurface soils collected from the study area

The obvious trend of Ni in subsurface soil in the study area was in the following order: open-burning site (29.65 mg/kg) > paddy field (9.67 mg/kg) > non-e-waste site (4.40 mg/kg) > groundwater-well area (3.86 mg/kg) > e-waste dismantling site (3.42 mg/kg). With respect to the statistical analysis, no significant difference of Ni in subsurface soil with the different sampling sites was observed. For the differences between Ni contents in surface and subsurface soil, the Ni in surface soil was significantly higher than in the subsurface soil at the groundwater well area and the e-waste dismantling site ($p<0.05$).

4.1.2.7 Pb concentrations in soils

A. Pb in surface soil

Figure 20 presents the Pb concentrations in surface soils in each sampling site. Pb ranged differently in the sampling sites. Pb ranged 12.12 to 13.22 mg/kg for the control site, 7.91 to 10.10 mg/kg for the non-e-waste site, 8.76 to 26.55 for the groundwater well area, 6.81 to 28.13 mg/kg for e-waste dismantling site, 4.35 to 44.03 mg/kg for paddy field, and 27.53 to 211.12 mg/kg for the open-burning site. For the reference site, Pb in surface soil was 10.27 mg/kg. The value of the reference Pb in the surface soil was close to Zarcinas et al. (2004) study. They found that mean of Pb concentration in the background area in Thailand, uncontaminated soils, was 11.0 mg/kg.

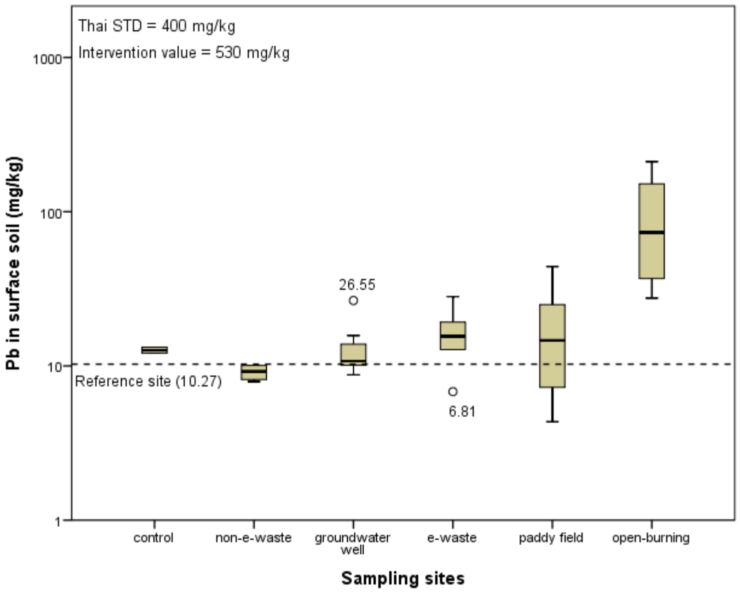


Figure 20 Pb concentrations in surface soils collected from the study area

Compared the Pb concentration to the reference Pb value (10.27 mg/kg), Pb in soils collected from the non-e-waste site was lower than the reference Pb, whereas Pb contents in some soil samples taken from the groundwater-well area, e-waste dismantling sites, and paddy fields were above the reference Pb. Additionally, Pb concentrations in all soil sampled from the open-burning site were higher than the reference Pb. Nevertheless, Pb in all samples did not exceed the intervention value and Thai standard set at 530 mg/kg and 400 mg/kg, respectively.

The particular trend of Pb in surface soil collected from various sampling sites were in the following descending order: open-burning site (99.19 mg/kg) > paddy fields (17.37 mg/kg) > e-waste dismantling site (16.50 mg/kg) > groundwater-well area (13.12 mg/kg) > non-e-waste site (9.13 mg/kg). Moreover, the statistical analysis also revealed that Pb in subsurface soil collected from the open-burning site was significantly greater than in the subsurface soil collected from the others sampling sites ($p<0.05$).

B. Pb in subsurface soil

Pb concentrations in subsurface soil collected from different sampling site were presented in Figure 21. Pb was varied among the sampling sites. Pb ranged from 14.34 mg/kg to 15.21 at control site, 7.29 to 12.79 mg/kg at non-e-waste site, 5.71 to 21.13 mg/kg at groundwater-well area, 4.73 to 9.89 mg/kg at e-waste dismantling site, 2.31 to 50.32 mg/kg at paddy field, and 14.79 to 2,700.77 mg/kg at open-burning site. For the reference site, Pb was 7.68 mg/kg in subsurface soil, which was similar to mean of Pb concentration in uncontaminated soils of Thailand (11.0 mg/kg) (Zarcinas et al., 2004). Based on the reference Pb contents in subsurface soil, Pb contents in all subsurface soil samples were above the reference, except for some soil samples collected from paddy-fields.

With the minimum and maximum concentration of Pb at open-burning site being 14.79 and 2,700.77 mg/kg, the Pb concentrations were about 2 to 352 folds higher than the reference Pb in subsurface soil, indicating an extremely contamination of Pb at the site where an open-burning of unwanted parts of electronic products, and municipal waste were operated. In addition, a dumping of smashed cathode ray tubes (CRT) screens, discarded printed circuit boards, and scraps of electronic parts after being dismantled were also carried out at the site. The result in this study was similar to Pradhan and Kumar (2014) study. They found high contaminated Pb in the soil (2,647.31 mg/kg) collected from e-waste dumping site at Mandoli industrial area, Delhi, India, where the e-waste dumping site was served for the dumping of e-waste after recycling activity, including printed circuit boarding, CRTs, cables and batteries. In addition, Kyere et al. (2016) also reported that the burning site of e-waste at informal e-waste recycling site in Agbogbloshie, Ghana was seriously contaminated with 2,666.4 mg/kg of Pb.

The concentration of Pb in subsurface soil with the mean of 700.55 mg/kg from open-burning site considerably exceeded both the Thailand standard value for residential and agricultural soils set at 400 mg/kg, and the intervention value set at 530 mg/kg. These results suggested environmental remediation is required. The obvious trend of Pb in subsurface soil was observed in the following order: open-burning site (700.56 mg/kg) > paddy fields (21.03 mg/kg) > non-e-waste site (10.04 mg/kg) > groundwater-well area (9.67 mg/kg) > e-waste dismantling site (7.99 mg/kg). Based on the statistical analysis, the difference of Pb in subsurface soil among sampling sites was not found, but the difference between Pb concentrations in surface and subsurface soil were significant in groundwater well area and e-waste dismantling site; the Pb in surface soil was significantly higher than in the subsurface soil ($p<0.05$).

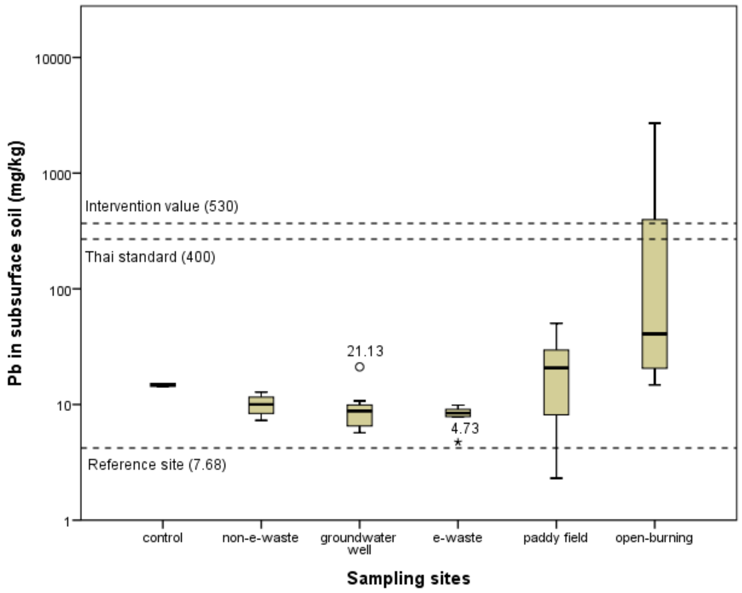


Figure 21 Pb concentrations in subsurface soils collected from the study area

4.1.2.8 Zn concentrations in soils

A. Zn in surface soil

Figure 22 shows Zn concentrations in surface soil taken from the sampling sites. Zn ranged from 5.68 to 6.37 mg/kg in control site, 3.34 to 26.94 in non-e-waste site, 7.20 to 160.05 mg/kg in groundwater-well area, 19.04 to 123.40 mg/kg in e-waste dismantling site, 1.84 to 27.23 mg/kg in paddy field, and 38.85 to 206.41 mg/kg in open-burning site. For the reference site, Zn was 6.88 mg/kg, and the Zn was lower than the global typical background concentrations, which were 10 to 100 mg/kg of Zn (Alloway, 2012).

Compared to the reference Zn in the study area, Zn contents in soil samples from the control site were over the reference value. Zn concentrations in some soil samples from the non-e-waste site and paddy-fields were above the reference Zn, while Zn of all soil samples collected from the groundwater well area, e-waste dismantling site, and the open-burning site was higher than the reference. However, Zn in all soil samples met the intervention value set at 720 mg/kg. The obvious trend of Zn in surface soil was observed in the following orders: open-burning site (126.09 mg/kg) > e-waste dismantling site (58.53 mg/kg) > groundwater-well area (43.67 mg/kg) > non-e-waste site (14.18 mg/kg) > paddy fields (10.65 mg/kg). In addition to this, according to the result of statistical analysis, Zn concentration in soil collected from the open-burning site was significantly higher than in the soil collected from the non-e-waste site, groundwater well area, paddy-field, and control site.

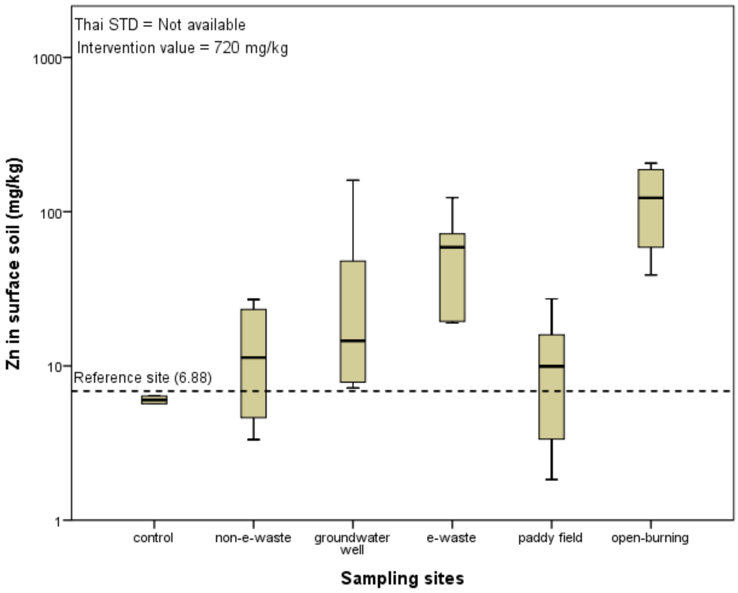


Figure 22 Zn concentrations in surface soils collected from the study area

B. Zn in subsurface soil

Zn concentrations in subsurface soil in the study area are presented in Figure 23; Zn varied widely between soils collected from different sampling sites. Zn ranged 6.03 to 7.17 in control site, 4.13 to 8.40 mg/kg in non-e-waste site, 3.67 to 56.67 mg/kg in groundwater-well area, 5.39 to 23.85 mg/kg in e-waste dismantling site, 0.65 to 24.08 mg/kg at paddy-field, and 34.62 to 1,735.86 mg/kg at open-burning site. The reference concentration of Zn was 5.64 mg/kg, which was lower than the Zn reported in other studies. Comparing Zn in subsurface soil collected from the sampling site to the reference Zn, Zn in all soil samples from the control site and open-burning area was higher than the reference level, whilst Zn in soil sampled from the other sampling site was partly above the reference.

Being 3-7 times higher than the reference Zn, the maximum Zn in subsurface soil at open-burning could be considered extremely contaminated; in addition, it also exceeded the Dutch intervention value for soil set at 720 mg/kg. The Zn analyzed in the burning site with the mean of 470.34 mg/kg was comparable with the Zn determined in the soil collected from e-waste recycling unit in Delhi, and backyard recycling sites slum areas in Bangalore, India (with the mean being 416.31 mg/kg, and 326 mg/kg, respectively). The recycling sites in these cities are served for an open-burning, shredding, and manual dismantling of e-waste, mainly for the burning of plastics coating in the electric wires, and disposing of the unwanted materials like CRTs (Ha et al., 2009; Pradhan & Kumar, 2014).

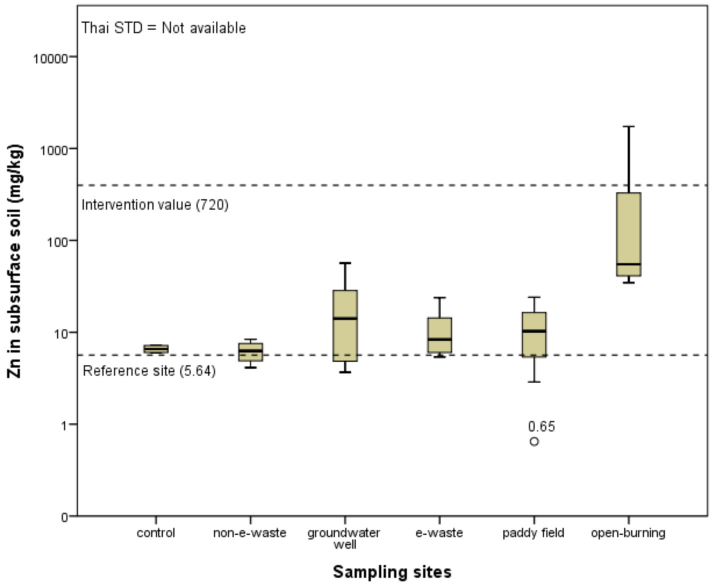


Figure 23 Zn concentrations in subsurface soils collected from the study area

The particular trend of Zn in the subsurface in the study area was noted as the following descending order: open-burning (470.34 mg/kg) > groundwater-well area (19.65 mg/kg) > e-waste dismantling site (11.58 mg/kg) > paddy field (10.73 mg/kg) > non-e-waste site (6.27 mg/kg). Nevertheless, the significant difference of Zn in the subsurface soils among the sampling sites, and that among the soil layers were not found. Heavy metal concentration in the soils in the study area were very varied among the sampling sites. The soil samples were taken from the reference site, control site, non-e-waste dismantling site, e-waste dismantling household site, groundwater well area, paddy fields, and open burning site.

To sum up section 4.1.2, as the reference site in this study was a forest and uncultivated soils, the site could be considered as an uncontaminated due to a minimal human activities and the distance was far from the study area over 5 km. Thus, metal concentrations in the soil from this area could represent as natural background values or reference values. According to Table 13, comparing the metal concentrations in the sampling sites with the reference values, all average metals measured from the soil at the non-e-waste site were less than or equal to the reference values, while metal contents in some of non-e-waste house were larger than the reference value. Similarly, for the groundwater well area, the metal contents in the soil in the area were slightly higher than their reference values, while Zn and Pb contents were highly greater than the reference values for about 6 and 7 times, respectively. The metal contents in soils from both non-e-waste house and groundwater well area that were higher the reference values could be attributed to the release of metals coming from e-waste dismantling activities carried out in the neighboring houses.

For the control site, which is located at a local school (5 km away from the e-waste dismantling village), the metal concentrations of Cd, Cr, and Cu were higher than the reference values for approximately 2 times, indicating that the metals in soil at the control site was slightly influenced by human activities. Based on field observation, even though there was no e-waste recycling activity in the area of the control site, the field of the school located near a rural highway and used to be a parking lot from time to time. In addition, the school was situated in a village where local residents always burn municipal solid wastes, and biomass in the open air which may be affected to elevate heavy metal contaminated in soils. Another possible cause, the traffic activity, like engine combustion and the traditional burning, have elevated the levels of Cd, Cr, and Cu in the soils at the control site via atmospheric deposition (Alloway, 2012).

For the paddy fields, the concentrations of As, Cd, Cr, Cu, and Pb exceeded those of the reference site for 2 to 3 times, suggesting that the paddy fields were contaminated with the metals. The level of paddy fields was lower and adjacent to the open-burning site that was a significant reservoir of leachate containing a high level of heavy metal contamination from OB. The other possible activities affecting the elevated concentration of these metals were rice farming itself, burning and dumping e-waste at the open-burning site. The concentrations of all metals in the paddy soils showed high variations. Since phosphatic fertilizers containing high amounts of As (2-1,200 mg/kg), Cd (0.1-170 mg/kg), Cr (66-600 mg/kg), Mn (40-2,000 mg/kg), Ni (7-38 mg/kg), Cu (1-300 mg/kg), Pb (7-225 mg/kg), and Zn (50-1,450 mg/kg) were applied in the paddy soils to gain high crop yields, the variation of the heavy metals in the paddy fields was probably due to the various levels of fertilizer loadings in the rice fields at the study area (Alloway, 2012). For this reason, the e-waste burning and dumping activities, and agricultural activities could be considered as sources of heavy metal contamination in the paddy fields.

In e-waste dismantling household area, the contents of Pb, Cu and Zn were higher than the reference metals for 2, 4, and 8 times, respectively; this means that anthropogenic activities, which are the dismantling of electronic devices and piling up of electronic products, might influence the contamination of Cu and Zn in the household area. Similarly, Fujimori and Takigami (2014) reported that Cu, Pb, and Zn in soil taken from an e-waste dismantling household workshop in Manila, the Philippines were excessive compared to the background heavy metal. The elevated levels of Cu and Zn in e-waste household soil are influenced by the deposition of dust and discarding of small fragments derived from the dismantling and piling up of e-waste onto soil surfaces. For the open-burning site, the contents of Cd, Pb, Zn, and Cu in the soils were higher than the reference values for 10, 10, 18.3, and 1,546 times, respectively, while As, Cr, and Ni contents were slightly greater than the reference values for 2.5, 3.5, and 5.4 times, respectively. The results suggested that the open-burning of electric wires and unwanted materials, smashing of CRT screens, and the dumping of unwanted materials lead to the serious contaminations of Cu, Zn, Pb, Cd, Ni, Cr, and As in the soils.

The trend of metal distribution in the study site was obvious that all metal contents in the surface and subsurface soil from the open-burning site were extremely high compared to other sampling sites. This was because of the uncontrolled e-waste recycling activities. The main activities being done in the burning site include (1) burning of electronic cables to easily recover coppers and unwanted materials (2) manual dismantling of CRT monitors and televisions to obtain metal components, and (3) dumping of unwanted materials. These crude techniques being conducted for over ten years have released a significant amount of metal-containing dust and particulate into the air, and they are later deposited onto the soil surface; as a result, elevated amounts of metal were added to the soils leading to the metal contamination in soil (Ackah, 2017; Zheng et al., 2016). Consequently, the open-burning site could be considered as a hot-spot for e-waste dismantling activities in the study area.

Metal contents in the soils from the paddy fields were secondly high compared to the burning site. As the paddy fields were located close to the burning site, the dust and particulates containing metals emitted from the e-waste processing activities could transport to the nearby area via deposition and atmospheric movement (Jinhui et al., 2011; Luo et al., 2011; Wong et al., 2007). The result of the metals contamination in the paddy field was consistent with the results previously reported by Jinhui et al. (2011) and Luo et al. (2011). They reported that metals released from open burning sites of e-waste in China might transport into the surrounding area, like vegetable gardens and paddy fields, through water irrigation and air deposition. In addition to this, previous studies also reported the consistent result that metal contamination in the e-waste recycling area could disperse beyond the e-waste dismantling site onto the surrounding area (Kyere et al., 2016). Open dumping, e-waste dismantling, and open-burning of e-waste emit leachates, coarse and fine particles, bottom ashes containing various heavy metals. These heavy metals could transport to soil media via multiple processes such as wet and dry atmospheric deposition, leaching by rainfall, adsorption-desorption (Li & Achal, 2020) (Moeckel et al., 2020).

Thai standard for residential and agricultural soils and the soil intervention value established by the Dutch ministry of housing were applied to assess the contamination levels of metals in soils in the study area. Cu concentrations with the means of 753 mg/kg in surface and 6,157 mg/kg in subsurface soils taken from the open-burning site exceeded the intervention value set at 190 mg/kg. Similarly, Pb in subsurface soils (700 mg/kg) from the open-burning site also exceeded both the Thai standard (400 mg/kg) and the intervention value (530 mg/kg). The maximum Zn contents in subsurface soil (1,735 mg/kg) collected from the open-burning site also exceeded the Zn intervention value set as 720 mg/kg. These results indicated that Cu, Pb, and Zn acquired from the e-waste open-burning and the open-dumping activities contribute to serious contamination at the open-burning site.

Based on the metal concentrations among soil layers at the e-waste dismantling sites, in the e-waste dismantling houses, the mean concentrations of all heavy metals were higher in surface layers (0-15 cm in depth) than in subsurface soils (15-30 cm). In the e-waste dismantling houses, the e-waste dismantling activities were continuously carried out on the surface area of the household backyard; therefore, the metals resulting from the dismantling activities are likely to contaminate in the surface soils. Likewise, heavy metals concentration were found to be higher in surface soil than in the deeper soils as the surface layer of soil were more actively influenced by anthropogenic supply, which was the e-waste recycling activity, compared to the deeper soils (Adelekan & Alawode, 2011; W. Zhao et al., 2015). In contrast, at the open-burning site, all metals contents were found higher in the subsurface soil (15-30 cm) than in the surface soils. The results might be due to the additions of new soils to the site. Addition of the new layer of soils was done on top of the surface soil in order to increase the disposal area served for the typical e-waste burning and dumping activities. Hence, high amounts of older residue and ash derived from the burning and dumping activities were buried in the subsurface layers. According to the mean concentration of heavy metals in the area where e-waste dismantling activities were conducted, all heavy metal concentrations in soil collected from an e-waste dismantling household area were lower than in soil from the open-burning site. This evidence suggested that an intensive e-waste burning activity contributed to the larger release of heavy metal than manually dismantling activities in household areas.

Heavy metal concentrations in soils in other e-waste recycling sites in China, India, and Ghana are listed in Table 13. Comparison heavy metal concentrations in the soil from an e-waste dismantling household area in this study to those in the other studies showed that the heavy metal contents in this study were lower than those in e-waste dismantling workshop in China, Vietnam, the Philippines, and Ghana (Fujimori & Takigami, 2014; Kyere et al., 2016; Lopez et al., 2011; Tang et al., 2010; W. Zhao et al., 2015). Likewise, the mean concentrations of all metals in surface soil from the open-burning site in this study were lower than those in China, India, and Ghana (Ha et al., 2009; Kyere et al., 2016; Luo et al., 2011; Wong et al., 2007). In China and India, apart from the typical open-burning activity, the extraction of precious metals by acid digestion, and the smelting for recovering metals were also performed in the sites. These intensive recycling activities could release higher amounts of metals to the receiving soils compared to the burning activities alone (Oguri et al., 2018; Suzuki et al., 2013; W. Zhao et al., 2015). Consequently, soils at the e-waste recycling sites in China and India were more contaminated with metals than in Buriram, Thailand (this study).

For the heavy metal contents in subsurface soil from the open-burning site, although there were not many studies focusing on the heavy metal concentrations in subsurface soils, the concentration of Pb in subsurface soil from the open-burning site in the present study was slightly higher than Pb in the residue from an open-burning site in Guiyu (Jinhui et al., 2011). Lastly, the comparison of heavy metal contents in paddy soils in the present study to other studies revealed that metal contents in this study were lower than those reported in other studies. According to the results in the present study, the lower levels found here might be caused by the e-waste activities operated in the Burriam e-waste site were mainly physical dismantling and separating, and burning activities, while other studies reported that e-waste activities mainly included acid digestion methods to extract precious metals out of the e-waste.

Table 13 Heavy metal concentrations in soils in this study and other studies

Author	Study area	Soil depth	site	As	Cd	Cr	Cu	Mn	Ni	Pb	Zn
This study	Buriram, Thailand	0-15 cm	E-waste dismantling house	6.76 (3.23-10.23)	0.40 (0.16-0.65)	12.56 (3.68-24.09)	28.23 (3.34-60.99)	103.93 (46.76-168.10)	5.56 (2.33-8.12)	16.50 (6.81-28.13)	58.53 (19.04-123.40)
			Non-e-waste dismantling house	8.30 (4.79-10.22)	0.35 (0.26-0.46)	11.93 (3.75-19.22)	10.20 (0.34-36.85)	64.95 (19.07-121.18)	3.73 (3.10-4.20)	9.13 (7.91-10.10)	14.19 (3.34-26.94)
			Groundwater-well area	7.89 (4.48-12.62)	0.37 (0.12-0.53)	9.46 (1.03-20.71)	4.09 (0.14-11.97)	97.83 (22.92-175.74)	4.98 (4.08-6.81)	13.12 (8.76-26.55)	43.67 (7.20-160.05)
			Open-burning site	16.41 (1.58-28.82)	1.24 (0.51-2.29)	31.15 (15.35-65.84)	753.32 (54.17-1725.45)	183.16 (47.55-365.83)	10.03 (5.17-17.89)	99.19 (27.53-211.12)	126.09 (38.85-206.41)
			Paddy field	11.95 (2.01-29.50)	0.68 (0.11-1.70)	19.69 (0.37-49.49)	19.51 (0.34-91.13)	104.42 (0.82-296.29)	7.08 (1.22-14.82)	17.37 (4.35-44.03)	10.66 (1.84-27.23)
			Reference site	9.73	0.45	11.61	6.02	304.14	7.09	10.27	6.88
		15-30 cm	E-waste dismantling house	5.77 (1.62-9.57)	0.29 (0.11-0.49)	11.23 (3.57-24.37)	3.14 (0.34-7.73)	68.75 (14.38-125.97)	3.41 (1.77-4.34)	7.99 (4.73-9.89)	11.58 (5.39-23.85)
			Non-e-waste dismantling house	9.07 (4.99-11.43)	0.42 (0.25-0.49)	15.19 (3.62-25.93)	2.81 (0.34-5.88)	73.75 (19.16-121.89)	4.40 (3.28-5.98)	10.03 (7.29-12.79)	6.27 (4.13-8.40)
			Groundwater-well area	5.92 (4.75–8.48)	0.30 (0.02-0.46)	7.60 (1.39-13.28)	1.27 (0.14-5.11)	97.63 (7.57-338.52)	3.86 (2.64-5.19)	9.67 (5.71-21.13)	19.65 (3.67-56.67)
			Open-burning site	18.91 (10.68 – 27.59)	3.46 (0.51-10.93)	34.89 (16.02-56.46)	6157 (10.30-24,431)	184.87 (86.50-338.13)	29.65 (5.02-94.17)	700.56 (14.79-2700)	470.34 (34.62-1736)
			Paddy field	18.03 (1.44-39.08)	0.89 (0.09-1.73)	22.61 (0.65-44.05)	13.10 (0.34-57.42)	171.75 (0.16-548.05)	9.67 (1.13-23.19)	21.03 (2.31-50.32)	10.73 (0.65-24.08)
			Reference site	7.56	0.35	9.99	190	108.35	5.50	7.68	5.64

Author	Study area	Soil depth	site	As	Cd	Cr	Cu	Mn	Ni	Pb	Zn
Ha et al. (2009)	Bangalore, India	Surface soil	Burning site	NA	0.385-38.9	73 (46-160)	592 (61.7-4790)	449 (286-849)	NA	297 (90.4-2850)	326 (126-2530)
Wong et al. (2007)	Guiyu, China	0-10 cm	Reservoir	NA	ND	84.1-150	29.5-42.7	NA	5.47-20.3	79.4-93.1	44.2-106
			Rice fields	NA	ND	81.9-509	29.9-240	NA	3.37-434	56.9-155	7.9-114
			Near the burning site 500 m	NA	ND	70.2-155	59.4-114	NA	4.04-68.7	97.8-123	6.01-42.4
			Open-burning site	NA	5.51-42.9	137-477	1374-14253	NA	85.2-722	856-7038	546-5298
W. Zhao et al. (2015)	Guangdong, China	0-20 cm	E-waste recycling area	4.7	0.128	58.1	50	45.3	57	77.5	102
Jinhui et al. (2011)	Guiyu, China	0-20 cm	Residue from an open-burning site	52.1	10.02	320	12700	500	1100	480	3500
			Paddy filed near open-burning site	23.09	1.6	48	140	200	30	62	130
			Paddy field near open-burning site	5.02	0.16	42	27	170	25	54	96
Luo et al. (2011)	Guangdong, South China	0-15 cm	Vegetable garden	NA	0.26–1.17	9.66–19	210–450	NA	7.04–10.3	73.3–134	92.4–142
			Paddy field	NA	0.04–1.43	10.5–24.1	40.1–260	NA	10.8–66	48.1–97	62.1–252
			Burning site	NA	3.05–46.8	23.6–122	1500–21,400	NA	12.2–132	629–7720	682–8970
			Deserted soil	NA	0.08–0.39	3.84–11.1	49.9–95.4	NA	4.58–19.7	47.2–60.2	44.5–72.3
			Pond area	NA	0.57–18.3	14–105	142–12,900	NA	18.9–44.9	37.9–7760	123–3800
Jun-hui and Hang	Taizhou, China	0-20 cm	Paddy fields without recycling activities	NA	0.09±0.20	9.78±7.97	41.39±18.81	177.58 ± 153.48	NA	28.13±19.29	74.88±62.77

Author	Study area	Soil depth	site	As	Cd	Cr	Cu	Mn	Ni	Pb	Zn
(2009)			Paddy fields without recycling activities	NA	0.17±0.02	10.40±8.71	46.93±17.89	461.02 ± 5.33	NA	49.63±20.55	94.64±20.89
			Paddy field with a few chipped circuit boards piled	NA	0.09±0.01	10.64±8.32	39.94±15.42	461.02± 10.67	NA	49.63±25.64	114.46±33.58
			Paddy field with a few chipped circuit boards piled	NA	0.11±0.13	21.66±6.02	46.63±1.20	435.25 ± 0.99	NA	65.16±2.40	138.0±7.51
			Paddy fields without un-dismantled computer piled	NA	0.33±0.08	33.49±28.74	51.19±10.64	508.52± 93.47	NA	67.04±19.90	145.45±29.35
			Paddy field with recycling operation (dismantled computer, washer ,TV)	NA	0.62±0.92	20.95±2.63	97.90±50.34	430.11± 79.76	NA	44.29±16.33	281.38±127.99
			PF surrounding a legal metal recycling center	NA	6.37±5.91	26.75±6.09	256.38±99.70	366.56± 57.75	NA	46.84±12.18	209.85±32.89
			no e-recycling history and no serious environment pollution	NA	0.15±0.01	6.33±1.03	32.08±2.11	324.81± 24.66	NA	33.45±1.33	111.99±37.77
Tang et al. (2010)	Wenling, China	0-30 cm	Simple household e-waste recycling workshops	NA	2.8	125.9	222.5	NA	71	200.1	221.2
				NA	7.9	65.9	361.7	NA	53.3	295.2	437.3
				NA	12.5	40.5	1641.3	NA	68.6	2374.1	518.7
Lopez et al. (2011)	Hong Kong	0-5 cm	Agricultural area	NA	1.34 ± 1	12.2 ± 2.4	28.4 ± 25.2	NA	NA	144.8 ± 47	121.5 ± 74.8
			E-waste storage	NA	0.08 ± 0.2	22.93 ± 1.6	11.5 ± 2.2	NA	NA	232 ± 32.5	114.6 ± 3.6
			E-waste dismantling workshop	NA	4.72 ± 2.9	1717 ± 1050	755.6 ± 611	NA	NA	1380 ± 1174	1717 ± 1050
			E-waste open-burning	NA	11.4 ± 3.9	272 ± 102.5	533 ± 267	NA	NA	3254 ± 1895	920 ± 489
Fujimori and	Manila, the Philippines		Informal recycling sites	3.7	NA	NA	680	950	47	800	900

Author	**Study area**	**Soil depth**	**site**	**As**	**Cd**	**Cr**	**Cu**	**Mn**	**Ni**	**Pb**	**Zn**
Takigami (2014)											
Pradhan and Kumar (2014)	Delhi, India	0-15 cm	Inside the recycling unit (S1)	12.85	1.14	83.57	6734.86	NA	146.5	2133.98	416.31
			E-waste dumping site (S2)	17.08	1.29	115.5	4291.61	NA	126.46	2645.31	776.84
			Arable land 50m away from S1	3.75	0.7	34.79	76.98	NA	44.67	40.28	90.28
			Arable land 100 m away from S1	ND	0.06	22	65.32	NA	35.11	29.61	68.36
			Arable land (adjacent residential area), 500 m away the S1	ND	0.04	16.35	63.45	NA	35.74	27.94	62.47
Kyere et al. (2016)	Agbogbloshie, Ghana	NA	Burning site	NA	10.1	171.5	2967.8	NA	21.8	2666.4	1887.2
			Dismantling site	NA	7.8	419.2	1643.5	NA	44.6	846.8	1939.2
			Residential site	NA	2.6	153.9	1354.6	NA	27.2	896.1	1170.4
			Recreational	NA	1.9	338.6	762.9	NA	14.8	355.2	700.3
Oguri et al. (2018)	Bui Dau, Vietnam	A few centimeters in depth	E-waste household garden soil	NA	1.70 (<0.03-3.54)	NA	771(61.9-4890)	550 (328-837)	NA	580(69.8-1980)	860 (109-1720)

“NA” means data is not available

4.1.2.9 Correlation between heavy metals in soils

Correlation between the studied heavy metals in the study area was examined by Pearson correlation analysis, and the correlation coefficients (*r*) were displayed in Table 14. Significant positive correlations were observed for all heavy metals, indicating the associations between heavy metals. Regarding the results obtained at $p<0.01$, the strong correlations with *r* higher than 0.7 were observed between multiple heavy metals. Cr had a strong association with As ($r = 0.80$); Mn showed strong correlation with As ($r = 0.73$). Cd had a strong correlation with Cu, Ni, Pb, and Zn, while Ni showed strong associations with Cu, Pb, and Zn. Additionally, Cu revealed strong correlations with Pb, and Zn, and Zn showed a strong association with Pb. The results indicated that the metals being closely associated with each other suggested their common origins.

Table 14 Pearson correlation coefficient (r) between heavy metals in the study area (n=72)

	As	Cd	Cr	Cu	Mn	Ni	Pb	Zn
As	1	.530**	**.807****	.238*	**.729****	.553**	.267*	.245*
Cd		1	.506**	**.945****	.455**	**.989****	**.954****	**.943****
Cr			1	.267*	.638**	.519**	.295*	.279*
Cu				1	.270*	**.930****	**.998****	**.988****
Mn					1	.519**	.291*	.280*
Ni						1	**.941****	**.926****
Pb							1	**.989****
Zn								1

**. Correlation is significant at the 0.01 level (2-tailed).

*. Correlation is significant at the 0.05 level (2-tailed).

4.1.2.10 Correlation between heavy metals and soil properties

Table 15 presents the result of Pearson correlation analysis between heavy metals and soil properties. The results showed significantly positive correlation between clay fraction and As, Cd, Cr, Mn, Ni, and Pb. Clay showed higher surface area for metal adsorption compared to sand and slit fraction. Additionally, organic matter, Fe-Mn oxides, and sulphides naturally occur in clay fractions (Yolcubal et al., 2004). The strong affinity of metals to the soil fractions is in the order of clay>silt>sand (Andersson, 1979). Regarding the study of Sungur et al. (2014), a positive correlation was found between the clay contents of soil and concentration of Mn, and it was reported that the amount of adsorbed metal is increased along with an elevated amount of clays. Thus, a high accumulation of metals in clay fraction is explained by the high surface area of clay for adsorption (Qian et al., 1996).

Soil pH positively correlated with Mn with the *r* of 0.357. pH is a master variable for controlling the solubility of heavy metals in soils; heavy metal adsorbs or precipitates as pH increase leading to the high total concentration of heavy metals (Rieuwerts et al., 1998). Similarly, Alloway (2012) reported that metals showed an increase in adsorption along with the increase of soil pH. This indicates the importance of specific adsorption in controlling metal ion solubility.

Table 15 Pearson correlation analysis between soil properties and total heavy metal concentrations

	As	Cd	Cr	Cu	Mn	Ni	Pb	Zn
pH	.180	.179	.175	.138	.357**	.187	.144	.197
ORP	-.120	-.138	-.139	-.108	-.077	-.119	-.115	-.135
OM (%)	.253	.249	.290	.089	-.067	.147	.134	.135
Clay (%)	.851**	.710**	.648**	.246	.600**	.735**	.381*	.049

**. Correlation is significant at the 0.01 level (2-tailed).

*. Correlation is significant at the 0.05 level (2-tailed).

4.1.3 Principal component analysis to identify possible sources of heavy metals

Regarding to section 4.1.2, there were large amounts of soil samples and multiples of heavy metal species, so an analysis aiming to reduce the data dimension into an associated set of heavy metals was essentially adopted in this study. Typically, either natural or anthropogenic processes attributed heavy metal contents in the soils. To clarify probably sources of heavy metals in the present study, principal component analysis (PCA) was then employed by utilizing the heavy metal concentrations in the soils. After the concentrations of eight heavy metals were statistically analyzed in PCA to reveal the associated group among heavy metals, the heavy metals could be extracted into two principal components (PC) that accounted for 92.58% of variance, as shown in Table 16.

Table 16 Loading of heavy metals in soil in PCA

Heavy metal	Principal component (PC)	
	PC1	PC2
Cu	**.993**	.098
Pb	**.990**	.129
Zn	**.987**	.109
Cd	**.917**	.386
Ni	**.897**	.427
As	.157	**.932**
Cr	.178	**.884**
Mn	.179	**.844**
% of variance	58.43	34.15
% accumulative	92.58	

Bold text represents the significant values of factor loadings (above 0.5)

The result means that approximately 92% of the heavy metal concentrations data set effectively represented the heavy metal in soils at this study site. The factor loading suggested being significant with the values of higher than 0.5 (Fujimori et al., 2012). PC1 took into the account for 58.43% of variance. In PC1, high loadings of Cu, Pb, Zn, Cd, and Ni were presented with the values of 0.99, 0.99, 0.99, 0.92, 0.90, respectively. Whereas, As, Cr, and Mn corresponded in PC2 with the loading of 0.93, 0.88, and 0.84, respectively. The PC1 and PC2 of heavy metals are also displayed in Figure 24. The heavy metals being categorized in the same PC were grouped together; the considerable distance between the two clusters showed the two most contrasting groups of heavy metals in the study areas.

PC1 consisted of Cu, Pb, Zn, Cd, and Ni, suggesting that these heavy metals were predominantly from the same anthropogenic source, which was the e-waste recycling activities in the study site. Various metals have utilized in electronic components. The examples of the metals presented in PC1 are used in multiple electronic products as follows: (1) Cu is used for electrical wires production, and conductivity of cathode ray tubes (CRTs), and printed circuit boards (PCBs), (2) Pb is commonly used as solders in PCBs, radiation shields in glass panels in CRTs, and transistor, (3) Zn is presented in CRTs as a coating for anti-corrosion purpose, (4) Cd is found in PCBs, batteries, phosphor emitters in CRTs, semiconductor chips, printer inks and toners, and Cd is also used as a plastic stabilizer, and (5) Ni is applied in batteries, semiconductors, PCBs, and CRTs (Kumar et al., 2017; Pagano et al., 2015; Perkins et al., 2014). During the dismantling and burning of e-waste, the heavy metal contained in e-waste could be released into the dismantling site, as well as the burning and dumping site. For this reason, PC1 represented the metals derived from e-waste recycling activities.

As, Cr, and Mn dominated in PC2, and these metals mostly derived from soil parent materials. As, Cr, and Mn can be considered as parent materials-derived metals in this study. Also, the consistent results were found in the study of He et al. (2017); they revealed that As and Cr in the e-waste recycling site in Qingyuan and Guiyu, China, were grouped in the same PC that was contributed by soil parent materials. In addition, Mn and Ba have the highest concentration in the earth’s upper crust, soil parent materials (Wedepohl, 1995).

PCA had been widely employed in some previous studies to identify possible sources of heavy metals contaminations in the areas having e-waste recycling activities. For instance, Han et al. (2019) studied heavy metal sources in soil samples collected from the e-waste recycling area in Qingyuan, China, by using PCA, and revealed that heavy metals were divided into two groups. The first group (PC1) consisted of Cd, Cu, Hg, Mn, Ni, Pb, Sb, suggesting that these heavy metals might have the same source. The second group (PC2) was corresponded with only As. These two groups of the heavy metals were concluded that PC1 reflected the impact of e-waste recycling activities, whereas PC2 was presented for geochemical sources of heavy metals, and the consequence of other anthropogenic activities.

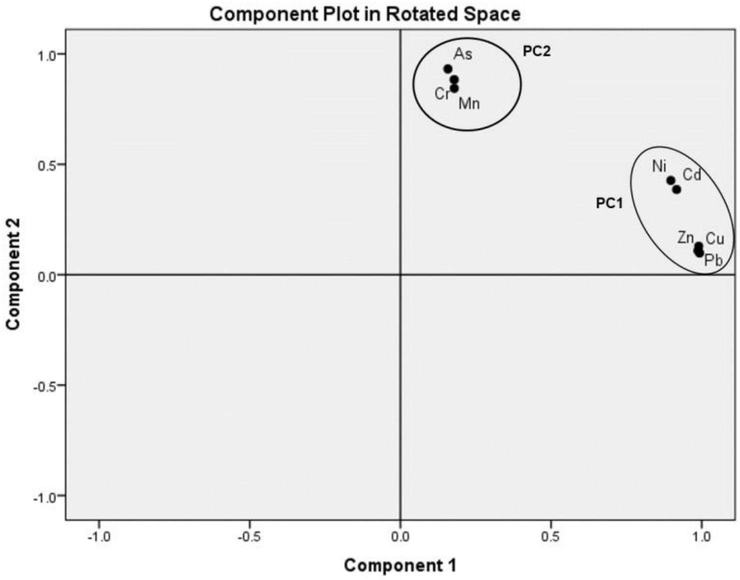


Figure 24 Loading plot of the principal component analysis (PCA) for heavy metal in soils

4.1.4 Heavy metals contribution in the soil taken from different land uses

After the total concentrations of all heavy metals in soil were discussed in section 4.1.2, the relative percentage contribution of each metal was calculated to reveal an abundance of each element and dominant elements in the soil of the study area, where different type of land uses were taken place. The contribution of heavy metals in surface and subsurface soil are presented in Figure 25 and Figure 26, respectively. The percent contribution of heavy metals in the study area is shown in appendix D.

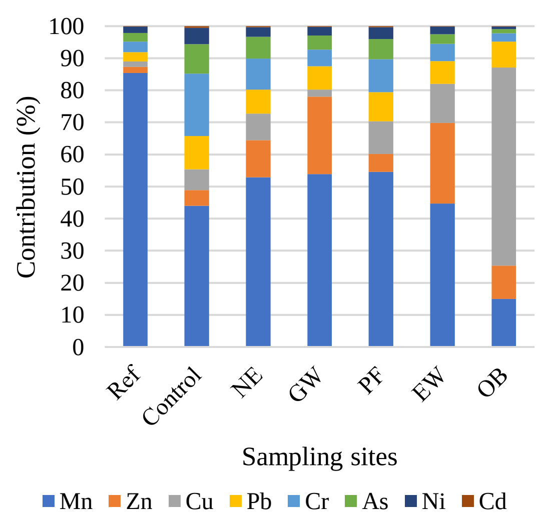


Figure 25 Heavy metal contributions in surface soil at the study area

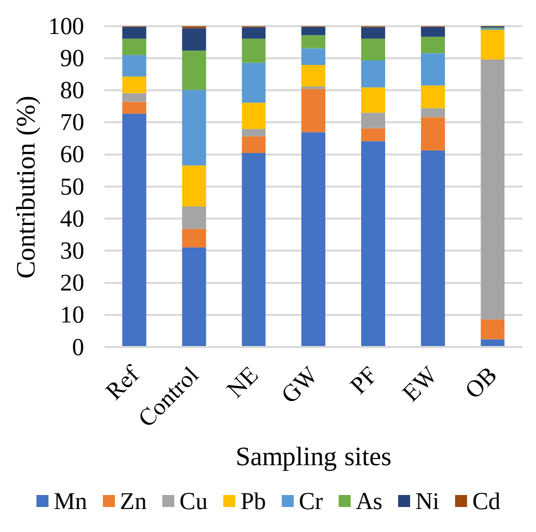


Figure 26 Heavy metal contributions in subsurface soil at the study area

Figure 25 presents the heavy metal contribution of each metal in surface soil. The metal contributions in the soil at different sites, including reference (Ref), control, non-e-waste houses (NE), e-waste dismantling houses (EW), groundwater-well areas (GW), paddy fields (PF), where without an intensive e-waste dismantling activity, were similar. The same pattern found at such mentioned sampling sites was that Mn accounted for the highest contribution with the values of 85% for Ref, 44% for control, 53% for NE, 45% for EW, 54% for GW, and 55% for PF. Besides, Mn in open-burning site (OB) was the second-abundant metal, accounting for 15%. According to the above results, Mn was the largest contributor in the sites without an intensive e-waste recycling activity, and Mn was the second-large contributor in the OB, where an intensive e-waste recycling activity was uncontrollably operated, like an open-burning of electrical wires. In addition, regarding the PCA results in section 4.1.3, reporting Mn was categorized as soil parent material-derived metal, Mn was geochemically presented in the site, and Mn could be considered as predominant metal in this study.

Considering the heavy metal contribution in surface soil from EW and OB where informal e-waste dismantling and recycling activities were taken place, apart from 45% of Mn contribution at EW site, the contributions of Zn, Cu, and Pb accounted for 25%, 12%, and 7%, respectively. Likewise, regardless of Mn mainly originated from soil parent materials, the large contributors in OB were also contributed by Cu (62%), Zn (10%), and Pb (8%). This contributions indicated that Cu, Zn, and Pb were the dominant species existed in soil at e-waste dismantling houses and e-waste burning sites.

According to Figure 26 showing the heavy metal contribution in subsurface soil from different sites, the contribution pattern and the main contributor of heavy metals was the same as found in surface soil; Mn had the largest contribution among all heavy metals in all sampling sites, except OB. Regardless of Mn contribution, the contributions of heavy metals in soil at EW and OB were slightly different. Cr, Zn, and Pb, accounting for 10%, 10%, and 7%, respectively, were the main contributors found at EW. On the other hand, Cu, Pb, and Zn were the main contributors at OB which accounted for 81%, 9%, and 6%, respectively. Based on the main contributions of heavy metals in this study site, it can be concluded that Mn was the most abundant metal in the site with no intensive e-waste recycling activities, while Cu, Zn, and Pb were the dominant elements in the e-waste recycling site. The high contributions of Cu, Zn, Pb found in OB and EW correspond with the results from the PC1, so it can be confirmed that Cu, Zn, and Pb accumulations in soils in the OB and EW were results of the e-waste dismantling and burning activities. However, Cr cannot be strongly concluded that it was the dominant metal in EW soil responding to the e-waste dismantling activities since the PC2 revealed that Cr in this study was considered as parent materials-derived metal.

With respect to Thongkaow et al. (2017) report, electronic products mainly imported to the study area were electric fans, CRT-based TVs and computers, washing machines, and refrigerators. After the electronic products were sorted, and dismantled, valuable materials, such as PCBs, plastics, wire cables, and copper, were obtained, while non-valuable materials were disposed. During the processes of manual dismantling, and sorting of the electronic products, fine fragments and scarps containing Cu, Pb, and Zn, which are the main elemental compositions in the imported devices, could fall onto the surface of soil leading to the contamination (Oguri et al., 2018). Similarly, Fujimori and Takigami (2014) revealed Cu, and Pb, and Zn - rich pattern of soil were found in the area of the informal e-waste dismantling workshops in the Philippines where CRTs, PCBs, cables were dismantled, and piled up.

In addition to manual dismantling of e-waste, an open-burning of insulated wires generated fly ash, and residual ash that contain high concentrations of heavy metals (Gullett et al., 2007). They reported emission factors of each element, which refers to the mass of the element per mass of the initial wires. The emission factors of Pb (964 mg/kg), Cl (785 mg/kg), Cu (106 mg/kg), and Zn (98.2 mg/kg) were relatively high among all elements. After the insulated wires were burned, the residual ash accounted for approximately 60% of the initial mass. Most elemental compositions in the residue ash were the contribution of Cl (293,000 µg/g), followed by Cu (47,000 µg/g), Pb (16,900 µg/g), Sb (883 µg/g), and Zn (764 µg/g), respectively. Subsequently, these residues containing large amounts of these metals may enter to soil compartment via wet and dry atmospheric deposition, causing the heavy metal pollution in the soil. As a result of their abundances in the cable wires, and the residue ash after being burned, Cu, Pb, and Zn have become the dominant species contaminating the OB, where e-waste workers regularly operated the open-burning of wires, and unwanted materials to recover coppers.

4.1.5 Enrichment factors of heavy metals in the study area

Enrichment factors (EFs) of metals are commonly used to comprehensively assess the heavy metal contaminations in soils as a result of human activities. The EF of each metal was calculated as expressed in Equation 3.9, and Mn was applied as a reference metal. Categorization of EF in accordance with the metals, and the mean EF values in each soil sampling site and their distribution are shown in Table 17 and Table 18 and Figure 27and Figure 28. The EFs of the heavy metals shown in Figure 27 and Figure 28 were varied among the sampling sites. The enrichment factor value of each sites was shown in appendix D.

According to Figure 27, considering the low quartile and high quartile (interquartile range), the EFs of surface soils in the control sites showed that the concentration of Ni and Zn indicated moderate to significant enrichment status, while As, Cr, Cd, Pb, Cu showed the EF value between 5 and 20, suggesting the significant enrichment. The EFs in the groundwater-well area (GW) showed minimal and moderate enrichment of Ni and Cr, whereas, the EFs of Cd and Cu showed the minimal to significant enrichment status. The concentrations of Pb and As showed the moderate and significant enrichment status in GW. The metals concentration in the surface soil of the non-e-waste sites (NE) revealed the minimal and significant enrichment status of Ni and Cu. In addition, the moderate and significant enrichment of As, Cr, Cd, and Pb were also found at the surface layer of NE. The EF of Zn showed the significant status in the NE. In the paddy fields (PF), EF values of all studied metals showed the moderate to significant enrichment in the surface soil.

Regarding the EF values of metals in the surface soil at e-waste dismantling sites (EW), wide ranges of EFs in each metal were observed. As showed the minimal enrichment, while Ni and Cd presented the minimal and moderate enrichment status in the EW. Moreover, Cr showed moderate enrichment status, whereas Pb showed moderate to significant enrichment status. Cu had moderate to very high enrichment status, and Zn caused significant and very high status in the EW. In the surface layer of the open-burning site (OB), Ni and As had moderate enrichment in the OB; Cr, and Cd showed the moderate to significant enrichment status in the OB. Conversely, Pb showed significant and very high enrichment. Cu showed extremely high enrichment status, and Zn showed very high enrichment status in the OB.

Figure 28 illustrates the Er of the metals determined in subsurface soil at the study area; those of all metals were found to be similar as in surface soil, and some were lower than that in surface soil. The result suggested that metal enrichment in the surface soil might occur due to the high retention at the surface soil. Similarly, W. Zhao et al. (2015) studied the enrichment of metals in soils at the e-waste recycling site in China, and the results showed the high retention of Ni, Cu, Cd, and Pb in the surface soil as the EF of these metals decreased along with the soil depth.

The metals found in each level of enrichment in the study area are presented in Table 17. In the control site, Cu, Pb, Cr, Cd, Zn, As, and Ni were significantly enriched in the surface soil with the mean EF between 5 and 20. Similarly, at the subsurface soil in the control site, Zn, and Ni were found with moderate enrichment levels, whereas the other metals were enriched in the significant status. The EFs in the control site were higher than 2, indicating anthropogenic sources rather than geogenic sources (Barbieri, 2016; Khalilova & Mammadov, 2016).

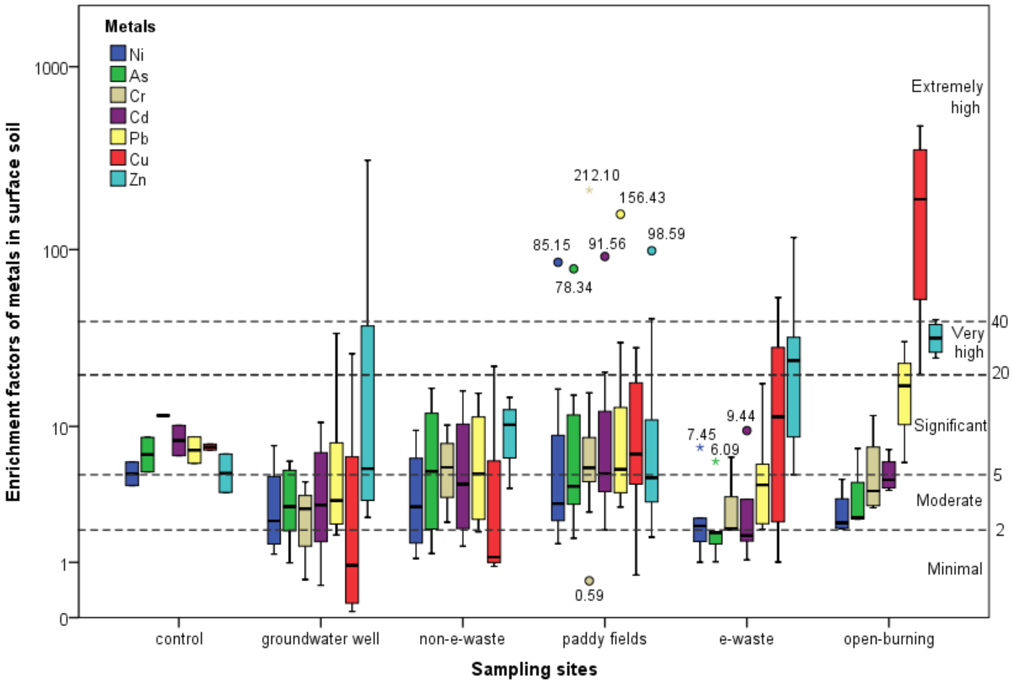


Figure 27 Enrichment factors of metals in surface soils in the study area

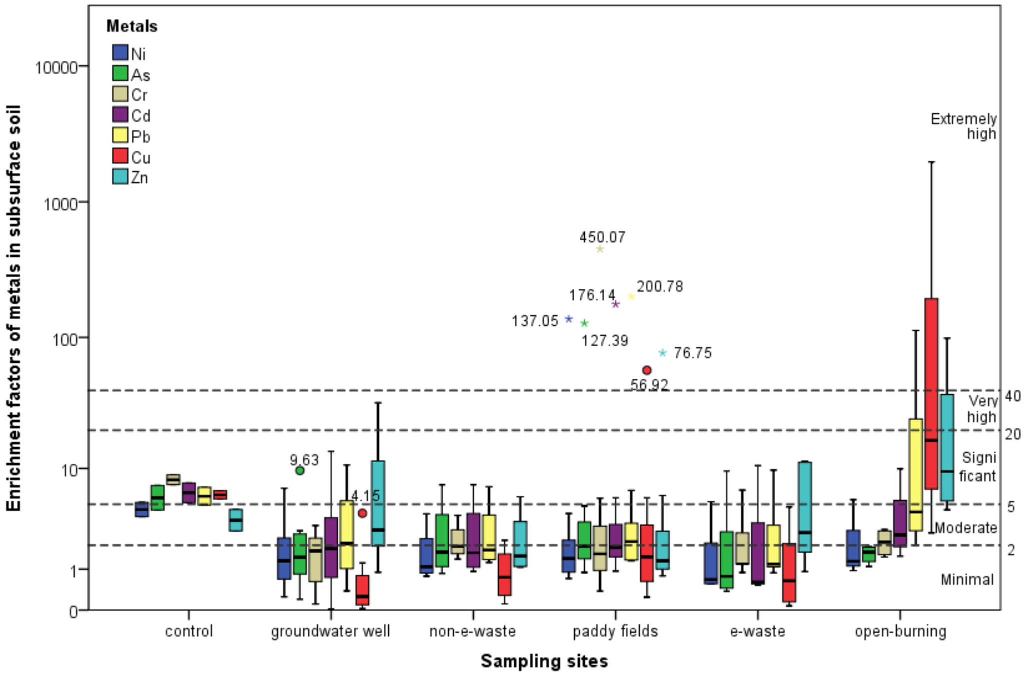


Figure 28 Enrichment factors of metals in subsurface soils in the study area

Regarding the metals enrichment in the groundwater-well area (GW) (Table 17), Cr, Cd, As, and Ni were moderately enriched in the surface soil. In addition, Cu and Pb were also showed significant enrichment status to the surface soil. As for the

EF level of the metals in subsurface soil at GW, Ni, Cr, and Cu were minimally enriched, while As, Cd, and Pb were moderately enriched in the subsurface layer. Moreover, Cu and Pb showed significant enrichment to the surface soil, and Zn also showed significant enrichment status in the subsurface soil.

Considering the metal enrichment in non-e-waste dismantling houses (NE), Ni showed moderate enrichment in the surface soil. Additionally, As, Cr, Cd, Pb, Cu, and Zn were significantly enriched in the surface soil. For those at the subsurface soil, Ni, and Cu had minimal enrichment status. Likely, As, Cr, Cd, Pb, and Zn showed the moderate enrichment status at the subsurface soil.

In paddy fields (PF), some surface soil samples were significantly enriched with Ni, As, Cd, Cu, and Zn. Conversely, subsurface soil samples were enriched by Ni, As, Cr, Cd, Pb, Cu, and Zn at the minimal and moderate level. For the metal enrichment in the soil at e-waste dismantling houses (EW), some surface soil samples were moderately enriched with Ni, Cr, and Cd. In addition, Cu was significantly enriched in the surface soil, and Zn was enriched at a very high level. Regarding the metals enrichment in subsurface soil at the EW, Ni, Cu, and Zn showed minimal enrichment at the site; As, Cr, Cd, and Pb were moderately enriched at the site. Zn was significantly enriched in the subsurface soil of EW. The enrichment factors of heavy metals in the e-waste dismantling site indicated the occurrence of Cu and Zn in the surface soil at the e-waste houses was highly enriched due to human activities like the manual dismantling and piling up of e-waste in the household area.

In the open-burning site (OB), Ni, and As were moderately enriched in the surface soil, and Cr, Cd, Pb, were significantly enriched in the surface soil. Zn and Cu showed very high and extremely high enrichment in the surface soil, respectively. With respect to the metal enrichment in the subsurface, Ni, Cr, and Cd showed a moderate enrichment level, and Pb and Zn showed a very high enrichment status in the subsurface. Cu had an extremely high enrichment status in the subsurface.

When compared among the sampling sites, very high enrichment of the metals was not presented in control, GW, NE, and PF sites. In contrast, Cu, Zn, and Pb showed very high enrichment in the EW and OD, and Cu showed extremely high in the OB. As the EFs of Cu, Zn, and Pb were very high and extremely high in soils from the e-waste dismantling houses and the e-waste open-burning sites in the comparison to soils in the other sampling sites. It can be indicated that Cu, Zn, and Pb occurrence in the soils could result from e-waste dismantling and e-waste burning activities in the study area.

With respect to the results of enrichment factor compared to the principal component analysis (PCA) (in section 4.1.3), the EFs of As and Cr showed the minimal enrichment level (being less than 2) in some samples taken from groundwater-well area, non-e-waste dismantling houses, paddy fields, e-waste dismantling houses, and open-burning sites. The result suggested that As and Cr enrichment in the mentioned sites might be derived from the soil parent materials. This result was consistent with the PCA result where As, Cr, and Mn were considered as parent materials-derived metals in this study.

Table 17 Categorization of EF in accordance with the metals found in the sampling sites

Sampling site	EF score and classification				
	< 2 (Minimal)	2 – 5 (Moderate)	5 – 20 (Significant)	20 – 40 (Very high)	> 40 (Extremely high)
Control					
-surface	-	Zn, Ni	**Cu***, **Pb***, **Cr***, **Cd***, **As***, **Zn***, **Ni***	-	-
-subsurface	-	As, **Zn***, **Ni***	Ni, **Cu***, **Pb***, **Cr***, **Cd***, **As***		
Groundwater-well area					
-surface	Ni, Cr, Cd, Cu	Cu, Pb, **Cr***, **Cd***, **As***, **Ni***, Zn	**Cu***, Zn, **Pb***, Cd, As	-	-
-subsurface	**Ni***, As, **Cr***, Cd, Pb, **Cu***	Ni, **As***, Cr, **Cd***, **Pb*,** Zn	**Zn***	-	-
Non-e-waste					
-surface	Ni, Cu	**Ni***, As, Cr, Cd, Pb, Cu	Ni, **As***, **Cr***, **Cd***, **Pb***, **Cu***, **Zn***	-	-
-subsurface	**Ni***, As, Cr, Cd, Pb, **Cu***, Zn	Ni, **As***, **Cr***, **Cd***, **Pb***, **Zn***	-	-	-
Paddy fields					
-surface	-	Ni, As, Cr, Cd, Pb, Cu, Zn	**Ni***, **As***, Cr, **Cd***, Pb, **Cu***, **Zn***	-	-
-subsurface	Ni, As, Cr, Cd, Pb, Cu, Zn	Ni, As, Cr, Cd, Pb, Cu, Zn	-	-	-
E-waste					
-surface	Ni, As, Cr	**Ni***, **Cr***, **Cd***, Pb, Cu	Pb, **Cu***, Zn	Cu, **Zn***	-
-subsurface	**Ni***, As, Cr, Cd, Pb, **Cu*,** Zn	**As***, **Cr***, **Cd***, **Pb***, Zn	**Zn***	-	-
Open-burning					
-surface	-	**Ni***, **As***, Cr, Cd	**Cr***, **Cd***, **Pb***	**Zn***	**Cu***
-subsurface	Ni, As, Cr	**Ni***, **Cr***, **Cd***, Pb	Cd, Pb, Cu, Zn	**Pb***, Cu, **Zn***	**Cu***

Remark: The classification of the EF is derived from the lower and upper of the box plot. "*" represents the enrichment level of the metal by mean EF.

Table 18 represents the mean value of enrichment factors of the heavy metals in the sampling sites. The mean EFs of all metals in the control site were greater than 2, indicating that the presence of the metals in the control site was probably due to anthropogenic activities. Similarly, the mean EFs for all metals in surface soil at GW, and those of Zn, Pb, Cd, and As in subsurface soil were higher than 2. These results suggested that the metals contents in the surface soil at GW and some metals in subsurface soil at GW might be the impact of human activities. In contrast, Cu, Cr, and Ni presenting the subsurface soil at the GW could come from the crustal soil as their EFs were lower than 2.

Table 18 Mean enrichment factor (EF) of the metals in the soils at all sampling sites (mean±SD)

Ele-ment	**Sampling sites**											
	Control		Groundwater-well area		Non-e-waste		Paddy fields		E-waste		Open-burning	
	Sur Face	Subsur-face	Surface	Subsur-face	Surf ace	Subsur-face	Surface	Subsur-face	Surface	Subsur-face	surface	Subsurface
Cu	7.49±0.44	6.05±0.69	5.73±9.66	0.82±1.39	6.39±10.66	0.98±0.93	11.15±9.45	6.43±15.98	19.53±22.34	1.54±1.96	223.11±194.3	501.29±978.49
Zn	5.29±2.10	3.65±1.17	63.17±114.56	8.31±10.75	9.84±4.43	2.49±2.25	16.16±28.12	8.12±21.66	37.39±45.68	5.59±5.27	32.71±7.15	30.83±45.42
Pb	7.28±1.91	5.95±1.45	8.05±11.08	3.66±3.89	7.2±6.27	2.97±2.75	20.96±43.37	19±57.26	6.44±6.54	3.21±3.75	17.86±10.16	30.82±54.60
Cr	11.61±0.79	8.11±1.04	2.64±1.34	1.61±1.12	5.92±3.25	2.31±1.14	23.38±59.56	39.3±129.36	3.21±1.93	2.51±2.39	5.61±4.06	2.19±0.73
Cd	8.39±2.49	6.39±1.71	4.35±3.61	3.33±4.43	6.73±3.32	2.9±3.00	14.34±25.07	16.75±50.21	3.46±3.45	3.12±4.31	5.11±1.49	4.15±3.90
As	6.9±2.37	5.84±1.97	3.47±1.91	2.47±3.02	7.4±6.94	2.9±3.01	12.05±21.39	12.64±36.16	2.49±2.05	2.78±3.89	3.7±2.43	1.59±0.38
Ni	5.14±1.27	4.53±0.93	3.34±2.46	1.99±2.13	4.21±3.73	1.76±1.56	11.79±23.56	12.88±39.11	2.94±2.58	1.84±2.02	2.82±1.24	2.25±2.16

In NE, the mean EFs of all studied metals in surface soil, and mean EF of Zn, Pb, Cr, Cd, and As in subsurface soil had the value being higher than 2. This suggested that the metals presenting in the NE could be a result of anthropogenic activities rather than natural sources. Conversely, EFs of Cu and Ni in the subsurface soil of NE were lower than 2, so these metals could be a natural presence. For the mean EFs in the paddy fields, all studied metals in the surface and subsurface soil showed the EFs that were higher than 2, indicating that the concentration of all studied metals might be affected by agricultural activities, such as fertilizer application in the rice fields (Payus & Talip, 2014).

The mean EFs for all metals in the surface soil, and those of Zn, Pb, Cr, Cd, and As in subsurface soil at the EW were higher than 2. The results indicating the enrichment of these metals in the EW could be a result of human activities, which were e-waste dismantling activities, and piling up of e-waste in the household area. Similar to the EFs found in the EW, mean EFs of all metals in the surface soil and subsurface soil at open-burning site, except for As in subsurface soil, were higher than 2. The results indicated that the enrichment of these metals could be derived mainly from the open-burning activities, and open-dumping of unwanted electronic materials, and scarps.

As a result of the metal enrichment presented, the highest level of enrichment for some metals was found in the open-burning area. At the same time, the surrounding site randomly showed the lower level of the enrichment with enrichment factors of metals being higher than 2. This indicated that anthropogenic activities were the main source of the enrichment of the metals in the study area. Besides, the results indicated that the open-burning site was the most contaminated area due to the intensive e-waste burning activity at the site. The results of this study were similar to previous studies. Pradhan and Kumar (2014) revealed that extremely high enrichment of heavy metals was found in soils at e-waste recycling sites, where manual separation, acid extraction, and heat-extruding of e-waste were carried out, while the soil from the area with no e-waste recycling activities had lower enrichment of metals. Also, Fosu-Mensah et al. (2017) calculated the enrichment factors of heavy metals in the soil at the e-waste area in Accra, Ghana, where an e-waste burning activities were operated. They reported that the e-waste area exhibited the highest value of enrichment factor compared to the non-e-waste recycling sites, indicating that the site served for the e-waste burning activities was the most contaminated in the study area.

To conclude the effects of e-waste dismantling activities on the heavy metal contaminations, the results from PCA, analysis of heavy metal contributions, and EFs were gathered and interpreted, as shown in Table 19. The overall results could show that e-waste dismantling activities influenced the heavy metal contaminations in the soils. The heavy metal contributions were consistent with the PC2; Mn was the most abundant heavy metal in all sites (except for OB), and Cr was abundant mainly in reference and control sites that were not influenced by human activities. Similarly, Mn and Cr were composed in the PC2, defined as the natural source-derived heavy metals. Consideration of the EF, EF of both EW and OB showed a higher level of Zn, Cu, and Pb (very high and extremely high level) compared to the other sampling sites, and Zn, Cu, and Pb were in PC1, which are e-waste derived heavy metals. The results indicated that the e-waste activities had caused the enrichment of Zn, Cu, Pb in the e-waste site.

Table 19 Comparative results of principal component analysis, heavy metal contributions, and enrichment factor

Site	PCA	HM contribution (HM abundant)	Enrichment factor			
			Moderate (2<EF<5)	Significant (5<EF<20)	Very high (20<EF<40)	Extremely high (EF>40)
Reference	**-PC1**: **E-waste derived-HM** Ni, Cd, Zn, Cu, and Pb **-PC2: natural source derived-HM** As Cr Mn	**Sur&sub**: Mn>Cr>Pb>As	-			
Control		**Sur&sub**: Mn>Cr>Pb>As	**Sur**: - **Sub**: Zn, Ni	**Sur:** Cu, Pb, Cr, Cd, As, Zn, Ni **Sub**: Cu, Pb, Cr, Cd, As		
NE		**Sur:** Mn>Zn>Cr>Pb **Sub**: Mn>Cr>Pb>As	**Sur:** Ni **Sub**: As, Cr, Cd, Pb, Zn	**Sur**: As, Cr, Cd, Pb, Cu, Zn		
GW		**Sur&sub**: Mn>Zn>Pb>Cr	**Sur**: Cr, Cd, As, Ni **Sub**: As, Cd, Pb	**Sur**: Cu, Pb **Sub:** Zn		
PF		**Sur:** Mn>Cr>Cu>Pb **Sub:** Mn>Cr>Pb>As	**Sur**: - **Sub**: -	**Sur:** Ni, As, Cd, Cu, Zn **Sub**: -		
EW		**Sur:** Mn>Zn>Cu>Pb **Sub:** Mn>Zn>Cr>Pb	**Sur**: Ni, Cr, Cd **Sub**: As, Cr, Cd, Pb	**Sur:** Cu **Sub:** Zn	**Sur**: Zn **Sub**: -	
OB		**Sur:** Cu>Mn>Zn>Pb **Sub**: Cu>Pb>Zn>Mn	**Sur**: Ni, As **Sub**: Ni, Cr, Cd	**Sur:** Cr, Cd, Pb **Sub**: -	**Sur**: Zn **Sub**: Pb, Zn	**Sur**: Cu **Sub**: Cu

4.2 Heavy metal fractionations in soils

The assessment of the mobility and potential bioavailability of the studied heavy metals was conducted by using sequential extraction based on the modified BCR method. In this study, the fractions of metals were classified into three fractions; F1 is an exchangeable and acid-soluble fraction; F2 is a reducible fraction, and F3 is an oxidizable fraction. The percentage distribution of metal fractions in the sampling sites was presented in Figure 29-Figure 44. The statistical analysis was conducted to compare differences of each fraction by one-way ANOVA with the confidential level at 95% using Tukey method, and the percentage of each fraction in wet and dry season was compared using T-test at 95% confidential level. The percentage distribution of heavy metal fractionations was shown in appendix E.

4.2.1 As fractionation in soil

Figure 29 represents the percentage of F1, F2, and F3 of As in surface soil at the sampling sites. The results showed that As mainly partitioned in F3, which binds to soil organic matter, in all sampling sites; more than 70% of As in control, GW, EW, and PF were bound to F1 in the control site, while more than 60% of As in OB were bound to F3. The results indicated that As in F3 was the dominant fraction in the study area. As in F3 is not a readily-mobile fraction, but potentially mobile fraction. Occurring in high levels of As and being in slightly acid conditions (pH 4.4 -6.58 in EW, and 5.03-6.06 in control) could pose a risk to soils in the study area by their increased mobility (He et al., 2017).

As presented in F3 is explained that As binds to the soil organic matter or organic materials by creating complexes (Wang et al., 2015). In addition, Alloway (2012) reported that As sorption to organic matter could be explained by the inner-sphere complexes of As with humic acid contained in soil organic matter. Considering F2 and F1 of As in the study site, As being in F2 and F1 was low in the study area, approximately 15-20% of F1-F2 combined was measured in control, GW, EW, and PF, while 25-30% of F1-F2 combined was found in OB.

Figure 30 represents the distribution of F1, F2, and F3 in the subsurface of the study area. The percentage of F1, F2, and F3 were similar to the results found in the surface soil. As presented in the subsurface soils were predominantly in F3, which is bound to soil organic matter in all sampling sites, except for the F3 at OB in the wet season. In wet season, F2 of As (64%) was the dominant fraction in the soil at OB, meaning that As associates with Fe oxy/hydroxides in soil by the adsorption process (Alloway, 2012).

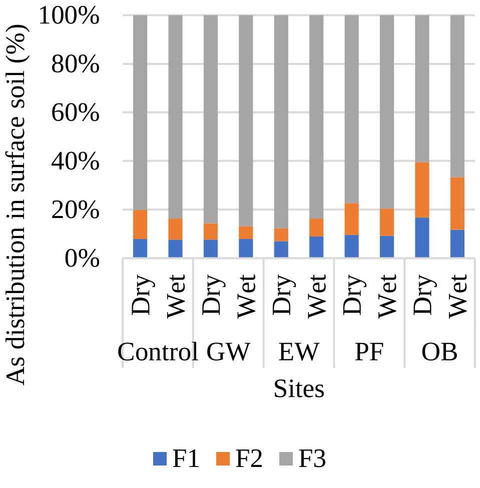


Figure 29 Distribution of As fractionation in surface soil

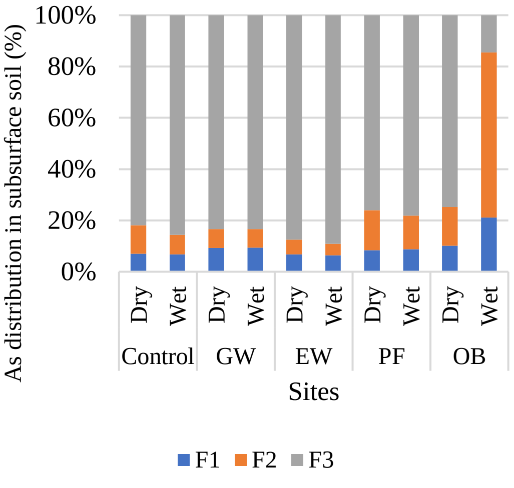


Figure 30 Distribution of As fractionation in subsurface soil

According to the statistical analysis, As presenting in F3 in all sampling sites, except for the OB, was significantly higher ($p<0.05$) than that in F2 and F1. It could be concluded that As bound to organic matter was the dominant fraction in the study area. Comparing each fraction among the sampling sites, the result showed that F1, F2, and F3 of As in OB were significant ($p<0.05$) higher than in the other sampling sites. Similarly, Damrongsiri et al. (2016) and Poopa et al. (2015) also reported that in heavy metal-contaminated sites, especially F1 and F2, are general forms of metals emitted from their sources, and then were settled and adsorbed in soil and also organic matter, clay, and Fe-Mn oxides.

Consequently, these non-residual fractions (F1, F2, and F3) could be higher in the contaminated open-burning soils. This can be concluded that As in both surface and subsurface soil presented at low risk due to low distribution of F1 and F2. As bound to F1 and F2 was slightly higher in OB than in the other sampling sites. According to soil pH, and ORP in the OB, soil pH ranged between 7.17 and 7.52 (neutral pH); specific adsorption strongly depends on soil pH. Also, at neutral pH, adsorption and complexation between metal and soil solid phase are major factors on the immobilization mechanism of metals (Rieuwerts et al., 1998). Consequently, As in the OB was more likely to be adsorbed and create complexes with soil solid phase than in the other sites.

Comparing each fraction among seasons revealed that, in control, F1 and F2 of As were significantly higher in dry than in wet season. Similarly, in PF, F1 and F2 of As were significantly ($p<0.05$) higher in dry than in wet season. The lower F1 was found in wet season could be explained by the dissolution of Fe and Mn oxides. The ORP in control and PF were reduced (from 301 mV to 253 mV in the control site, and from 311 mV to 293 mV in PF) in the wet season; under this condition, the dissolution of Fe and Mn oxides attached on the soil surface occurred. Thus, the adsorbed metal on the Fe-Mn oxides could be released and more exchangeable, so F2 in wet season was lower than in dry season (Isimekhai et al., 2017; Rieuwerts et al., 1998; W. Zhao et al., 2015). In contrast, in GW and OB, F3 of As in wet season was significantly ($p<0.05$) higher than in dry season.

4.2.2 Cd fractionation in soils

Figure 31 shows the distribution of Cd fractionation in surface soil in the study area. Cd showed the highest percentage for a reducible fraction (F2) for the soil samples collected from control (75% in dry and 64% in wet season), GW (64% in dry and 60% wet season), PF (64% and 70% dry and wet season, respectively). This indicated that fraction of Cd was mainly presented to Fe-Mn oxyhydroxides fractions in soils, which typically stable under normal environmental conditions, and is able to mobilize in soils under acidic conditions (Borgese et al., 2013). The relatively high level of Cd in the F2 might be because of the tendency of Cd to adsorb onto Fe-Mn oxides and coprecipitate with metal oxides (Borgese et al., 2013; Favas et al., 2011).

In contrast, Cd showed the highest distribution in F1 for EW in the dry season (40.52%), OB (50, 52 %, respectively). However, Cd showed comparable percentages of F1 and F2 in surface soil at EW, OB. Thus, the most potential bioavailability of Cd in the EW and OB was higher than in the other sites. It also could pose a risk to the environment more than other heavy metals. The obtained results in the study area were consistent with Isimekhai et al. (2017) research; they studied heavy metal speciation in soils at e-waste dismantling and recycling sites in Nigeria and reported that Cd had the highest percentage to associate with an exchangeable fraction (F1). Similarly, Cd showed the same characteristic of association with acid soluble fraction (F1) in the soil at e-waste recycling areas in Qingyuan, China (He et al., 2017). Since Cd highly partitioned to F1 was found in EW and OB, high mobility and bioavailability of Cd might be influenced by e-waste dismantling and burning activities.

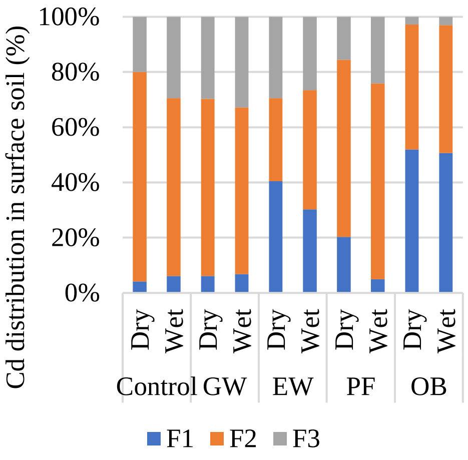


Figure 31 Distribution of Cd fractionation in surface soil

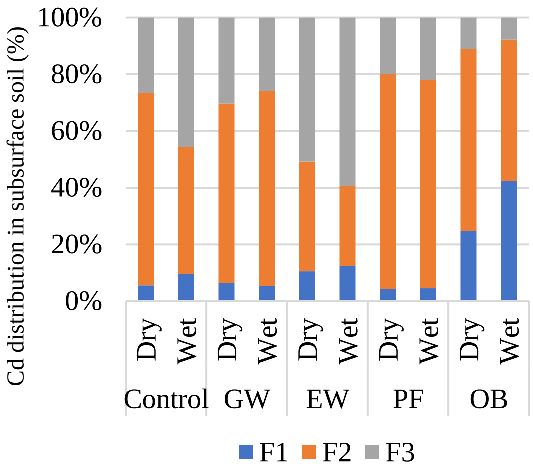


Figure 32 Distribution of Cd fractionation in subsurface soil

Figure 32 represents the Cd fractionations of subsurface soil in the study area. Cd in the subsurface soil of the control site was mainly associated with F2 (68%) in dry season, and Cd was found binding to F2 (44.8%) and F3 (45%) in comparable level in wet season. In addition, Cd in GW (63% in dry and 68% in wet season), PF (75% in dry, and 73% in wet season), and OB (64% in dry season and 50% in wet season) highly partitioned to F2. Conversely, Cd in EW mainly resided to F3 (50% for dry and 59% for wet season), meaning that Cd in EW in subsurface soil was likely associated to the organic phase, and this fraction is reported to be relatively stable in the natural environment; however, under oxidizing condition, organic phase could be decomposed, so the metals can be released from the organic phase to the soil solution. (Borgese et al., 2013; Rieuwerts et al., 1998)

Cd was likely to be mainly formed reducible (F2), followed by oxidizable (F3), and an exchangeable fraction (F1) in control and GW. In addition, Alloway (2012) reported that the main Cd adsorbents in soils are Fe-Mn oxyhydroxides, soil organic matter, and clay minerals; as a result, Cd was more largely partitioned to F2 and F3 rather than F1. Moreover, percentages of F1, F2, and F3 were reported to be significantly highest in OB among all sampling sites. Comparing Cd fraction among seasons, Cd in F2 at the control site was significantly higher in dry seasons rather than wet seasons. According to Ahmadipour et al. (2014) study, Cd is largely associated with amorphous Mn hydroxides and amorphous Fe hydroxides (F2). Under the reducing condition, Fe-Mn oxides are potentially dissolved, leading to the release of Cd adsorbed on Fe-Mn oxides. Consequently, Cd became unstable, and mobile under reducing condition which was easily released into the aqueous environment and potentially enter food chains (Ahmadipour et al., 2014; Borgese et al., 2013).

4.2.3 Cr fractionation in soils

Figure 33 presents the percentage of Cr fractionation in surface soil. Cr fractionation in control and GW have highly presented in F2; Cr in F2 at the control site accounted for 85% in dry season and 52% in wet season, respectively. F2 of Cr in GW was comparable between dry season (50.6%) and wet (48.5%) season. For Cr in EW, Cr was mainly presented in F2 (40.5%) in dry season, mostly presented in F3 (55.3%) in wet season. Regarding the Cr form in PF, Cr was mostly found in F3 (48.4%), whereas F2 (51.1%) was the main fraction for Cr in wet season. For Cr fraction in OB, Cr was highly bound to organic matter and sulfides (F3) (62.4% in dry and 63.0 % in wet season).

Figure 34 represents Cr fractionation in subsurface soil in the study area. Cr was mostly bound to F2 in control (87% in dry season, and 63.3% in wet season). In GW area, Cr was mainly partitioned to F2 (47.8%) in dry season, while Cr showed a comparable level of F2 (43.5%) and F3 (41.7%) in wet season. In contrast, Cr distribution in EW and PF showed similar contribution that Cr was mainly F3; F3 of Cr in EW accounted for 42.0% for dry season, and 49.4% in wet season. F3 of Cr in PF was 41.3% in dry season and 53.1% in wet season. For Cr fractionation in OB, F3 (90.8%) was the dominant fraction of Cr presented in the dry season, but F2 (49.5%) accounted for the dominant fraction of Cr in wet season.

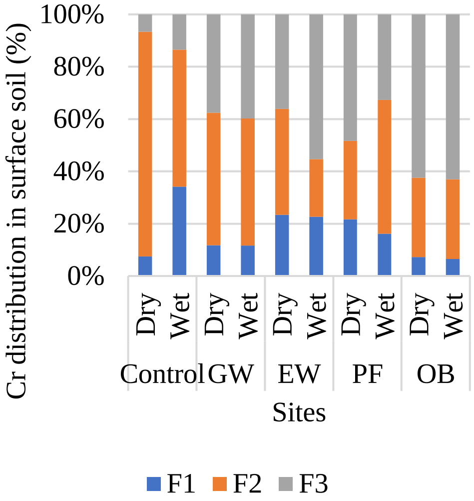


Figure 33 Distribution of Cr fractionation in surface soil

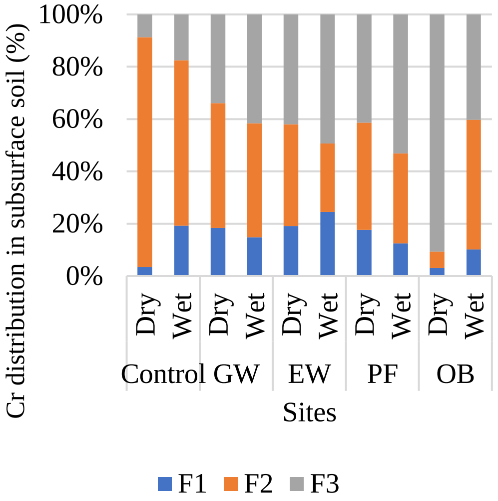


Figure 34 Distribution of Cr fractionation in subsurface soil

The statistical analysis showed that F2 of Cr in the control site was significantly higher than F1 and F3 ($p<0.05$), respectively. In contrast, F3 was significantly higher than F1 and F2 in EW. F1 found at PF and GW was significantly lower than F2 and F3 found in the sites. Comparing the Cr fractions among the sampling sites revealed that F1, F2, and F3 had the highest amounts in OB. The differences of the metal fractions among seasons revealed that Cr presenting in F1 at the control site was significantly higher in dry season than in wet season ($p<0.05$). Similarly, in PF, F1 of Cr in dry season was significantly higher than in wet season ($p<0.05$). Conversely, F3 in GW showed a higher significant level in wet season than dry season. The F3 was higher found in wet season could be explained by that as redox potential was slightly dropped in wet season, sulfide-containing phases could be precipitated in soils, and then the released Cr from the soils could be co-precipitated with the sulfide-containing materials, which is considered as oxidizable form (F3) (Lee, 2006).

4.2.4 Cu fractionation in soils

Cu fractionation in surface soil at the study area is illustrated in Figure 35. Cu in control is predominantly bound to F2 (85% in dry season, and 83% in wet season). Similarly, Cu at GW is mostly bound to F2 (54%) in dry season, while mostly bound to F1 (47%) in wet season. Regarding the Cu fraction in EW, F1 and F2 showed comparable percentages in dry season, while F1 (63%) showed a relatively high percentage in wet season. Similarly, F1 (41.4%) also showed a high contribution to the soil at the OB in dry season. This indicated that these exchangeable and acid-soluble (F1) of Cu in EW and OB could be readily mobilized and facilitated to the surrounding area, such as the paddy fields and the household vegetable gardens, which eventually could be taken up by plants, and accumulated in edible parts of plants (Luo et al., 2011). Cu in PF was mainly bound to F2 with the percentage of 54.1% in dry season and 58.3% in wet season.

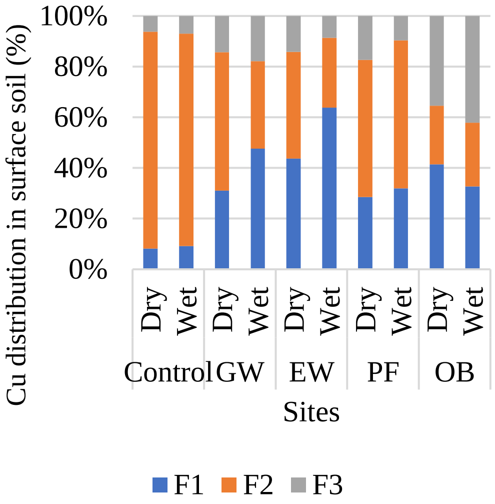


Figure 35 Distribution of Cu fractionation in surface soil

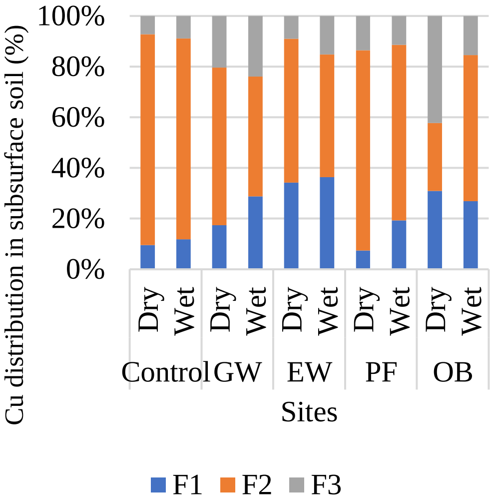


Figure 36 Distribution of Cu fractionation in subsurface soil

Figure 36 illustrates the Cu fractionation in subsurface soil in the study area. Cu in subsurface layers at controls, GW, EW, PF mostly partitioned to F2; on the other hand, Cu fractions in OB showed the highest percentage for F3 in dry season (42.3%), and F2 in wet season (57.6%). According to the statistical analysis, Cu fraction in the control site was in the order of F2> F3>F1. For EW, Cu in an oxidizable form (F3) was significantly higher than in F1 and F2, respectively. For PF and GW, Cu in F2 >F3 > F1, respectively. This can be implied low risk by lower leachable fractions. According to Alloway (2012), after soils draining and becoming drier through evaporation, ORP would rise, and Fe/Mn oxides re-precipitate, which lead to re-adsorb of Cu on Fe/Mn oxides. Similarly, Lee (2006) also reported that after the flooding periods in paddy fields, dissolved or released metals in soil solutions could be re-adsorbed either by Fe/Mn oxides or sulphide containing solid phase. Consequently, Cu bound to Fe/Mn oxides (F2) was high in the PF in the tiller stage (after flooding stage), which was the period the paddy soil samples were collected in the wet season.

The statistical results indicated that Cu in the study area most likely to associate with Fe-Mn oxyhydroxides (F2) and organic matter (F3) rather than F1. The result is in agreement with Mclaren and Crawford (1973) study; they studied the fractionation of Cu and found that after Cu^{2+} is brought into soils, Cu^{2+} will bind with varying affinities in soil constituent in the order of Mn oxides> organic matter> Fe oxides > clay minerals. Moreover, organic matter was reported to dominate Cu adsorption in the soil and being responsible for retaining adsorbed Cu (Alloway, 2012). In addition, a considerable amount of Cu was predominantly bound to F2 and F3, and Cu was bound to functional groups that occurred in humic and fulvic acid contained in soil organic matter as complexes. Cu typically forms a link to soil organic matter when Cu is applied to soils having high organic matrices, such as biosolids and manure (Orroño & Lavado, 2009). The differences of Cu fractions among the sampling sites revealed that all fractions of Cu (F1, F2, and F3) in OB were significant than those in the other sampling sites, and the differences of Cu fractions among seasons showed that organic-matter bound Cu in wet season in GW was significantly higher than in dry season ($p<0.05$).

4.2.5 Mn fractionation in soils

Figure 37 shows the distribution of Mn fractionation in surface soil in study area. Mn was predominantly associated with Fe-Mn oxides (F2) in the control site (56% in dry season, and 52.9% in wet season) and PF (67.4% in dry season, and 68.2% in wet season). In contrast, Mn was mainly associated with F1 for GW (74.6% in dry season, and 72.1% in wet season), EW (56.6% in dry season, and 70.2% in wet season). For the Mn fractions in surface OB, F2 was the main fraction of Mn with the percentage of 53.1% in dry season, while F1 showed the highest fraction in wet season with the percentage of 52.8%.

Figure 38 presents Mn fraction in subsurface soils at the study area. For the control site, Mn fractions showed the greatest percentage of F1 for dry season (55.4%), and was mainly the percentage of F2 (66.6%) in wet season. For the Mn fractions in GW, the contributions of F1 and F2 were comparable in both dry and wet season. For Mn fractionation in EW, exchangeable Mn (F1) was the dominant fraction (77.9% in dry season, and 64.7 % in wet season). The high proportion of F1 was in agreement with Ashraf et al. (2012) research, which reported that Mn is weakly sorbed and retained on the soil surface by weak electrostatic interactions, and Mn could be released by ion-exchange processes and dissociation of Mn-carbonate phase.

Moreover, Mn could possibly be released into the environment when the environmental conditions become more acidic. In contrast, Mn was mostly found F2 in the PF (86.9% in dry season, and 79.1% in wet season), and OB (56.2% in dry season, and 62.5% in wet season). Likewise, Borgese et al. (2013) and Pueyo et al. (2003) reported that F2 accounted for Mn that is primarily associated to Fe-Mn oxides in soils, and in slightly contaminated soils, Mn is mostly associated with Fe-Mn oxide bound form (F2) as Mn -oxides.

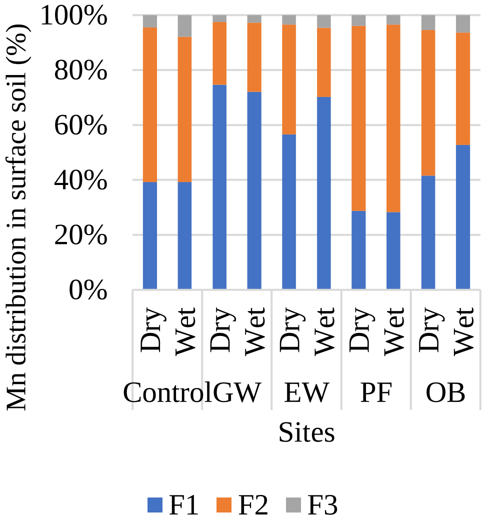


Figure 37 Distribution of Mn fractionation in surface soil

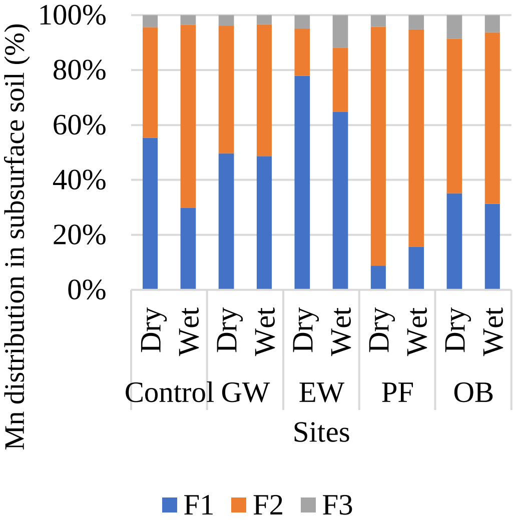


Figure 38 Distribution of Mn fractionation in subsurface soil

F1 and F2 of Mn were significantly found in greater distribution than F3 in all sampling sites. Obviously, Mn fractions of OB majorly found a higher proportion of F1 compared to other sites. F2 of Mn in PF was significantly higher than in control, and EW, while F2 in OB was significantly higher than in the other sites. Similarly, F3 in PF and OB were significantly higher than in control, EW, and GW. The Mn fractions among seasons were concluded that that Mn in a reducible form (F2) in dry season was higher than in wet season. While Mn in an exchangeable and acid soluble form (F1) at GW in wet season had a significantly higher amount than in dry season. However, Mn is generally contained in the parent material of soils and it also defines as essential micronutrients for plant growth. For Mn in GW, it potentially affected increase the water hardness. Thus, the results indicated that a higher distribution of F1 did not show the severe risk compared to other heavy metals (Alloway, 2012).

4.2.6 Ni fractionation in soils

Figure 39 illustrates Ni fraction in surface soil at the study area. Ni was mainly contained in the oxidizable fraction (F3) of the samples taken from the control site (47.5% in dry season and 47.9% in wet season), and PF (56.3% in dry season and 54.4% in wet season). In contrast, Ni was predominantly contained in the exchangeable fraction (F1) of soil at GW, and EW. For Ni fraction in OB, Ni showed a comparable percentage in all fractions.

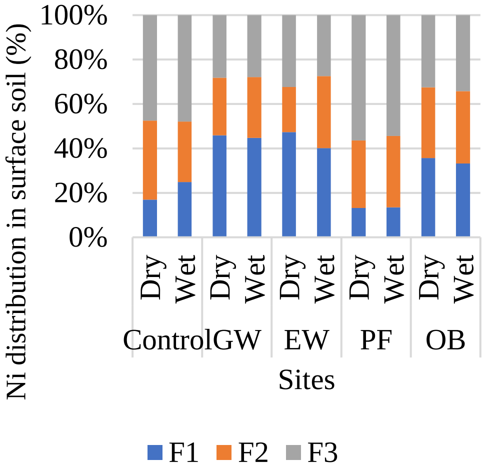


Figure 39 Distribution of Ni fractionation in surface soil

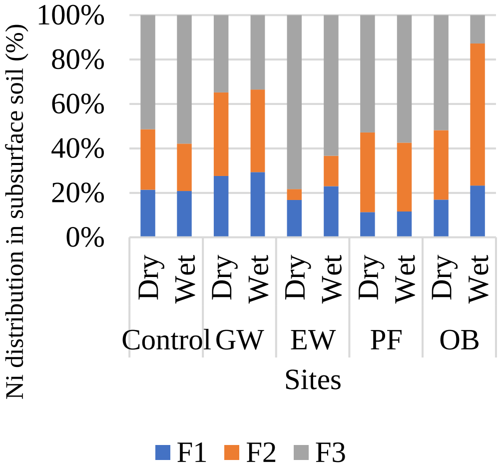


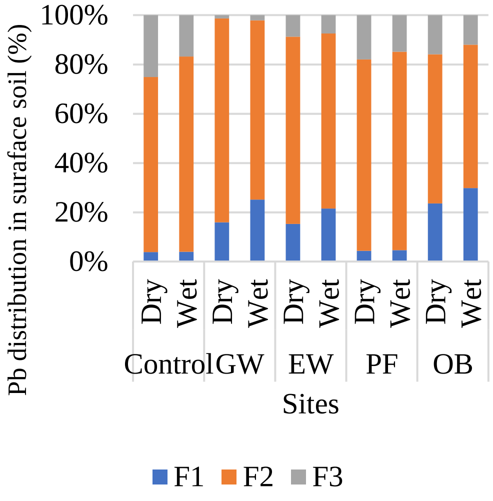
Figure 40 Distribution of Ni fractionation in subsurface soil

The percentage distribution of Ni fraction in subsurface soils is presented in Figure 40. Ni was mainly associated to soil organic matter (F3) in the control site (51.4% for dry season, and 57.9% for wet season), EW (78.3% in dry season, and 63.3% in wet season), PF (52.8% for dry season, and 57.4% in wet season), and OB (only in dry season for 51.8%). Ni was mainly bound to reducible phase (F2) in GW and OB (only in wet season).

The statistical analysis revealed that the significantly decreasing level of Ni fraction was in the order of F3>F2>F1 in control and PF ($p<0.05$). This could be because Ni was mainly associated with soil organic matter and sulfides by forming complexes (Rieuwerts et al., 1998; Wang et al., 2015). In addition, Gonnelli and Renella (2013) studied the chemical behavior of Nickel in soils and found that Ni has a relatively high affinity for organic matter. Ni (II) is bound to organic ligands, humic substance or particulate organic matter to generate Ni (II) organic complexes. Additionally, F1 of Ni in OB was significantly higher ($p<0.05$) than in the other sampling sites, which indicated that Ni in the OB was readily available and easily mobile to the surrounding area, leading to environmental risk in the area. Moreover, the combination of F1 and F2 in GW and EW showed a higher proportion compared to F3, suggesting that Ni could pose a risk to the ecosystem in GW and EW due to its potential mobility and bioavailability.

4.2.7 Pb fractionation in soils

Figure 41 represents the distribution of Pb in surface soil in the study area. The distribution of Pb was observed; Pb was predominantly bound to a reducible fraction (F2) in all sampling sites. Figure 42 illustrates Pb fractionation in subsurface soil in the study area. The percent distribution of Pb in subsurface presented a similar tendency as in surface soils; Pb in subsurface also mainly presented in F2 in all sampling sites.



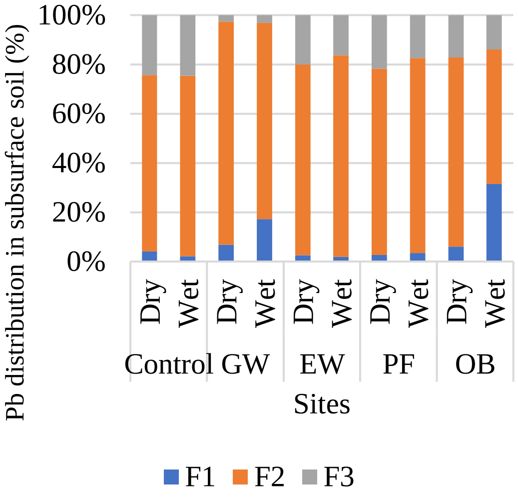


Figure 41 Distribution of Pb fractionation in surface soil

Figure 42 Distribution of Pb fractionation in subsurface soil

The statistical analysis showed that Pb showed significant Fe/Mn oxides fraction (F2) in all sites. The differences of the Pb fraction among the sampling sites showed that F2 and F3 in OB were significantly higher ($p<0.05$) than in the other sites; the results showed consistency with Borgese et al. (2013) reporting that the relatively high portion of Pb was bound to reducible form (F2) because of the tendency of Pb to adsorb onto Fe, Mn oxides, and subsequently Pb would coprecipitate with metal oxides (Favas et al., 2011). Wang et al. (2015) also reported that Pb bound to Fe-Mn oxide was the dominant fraction in soil. Therefore, in the study area, Fe-Mn oxides could be a factor controlling the mobility and bioavailability of Pb in the study area. In addition, He et al. (2017) and Yu et al. (2011) reported that Pb (45.5%) was mainly bound to reducible fractions, indicating that Pb could potentially mobile and bioavailable in the soils environment.

4.2.8 Zn fractionation in soils

Figure 43 presents the distribution of Zn fractionation in surface soil at the study area. The result showed that Zn was mainly found in F1 in the control site, GW, EW, and OB. Zn was bound to F1 with the percentage of 36.5% in dry season and 48.4% in wet season in the control site. F1 showed 77.6% in dry season, and 77.1% in wet season in GW. Similarly, Zn formed to F1 for 60.4% in dry season, and 64.7% in wet season at EW. In OB, Zn was F1 for 68.8% in dry season and 72.9% in wet season. Conversely, Zn in PF was mainly F2, accounting for 38.5% in dry season, and 40.9% in wet season.

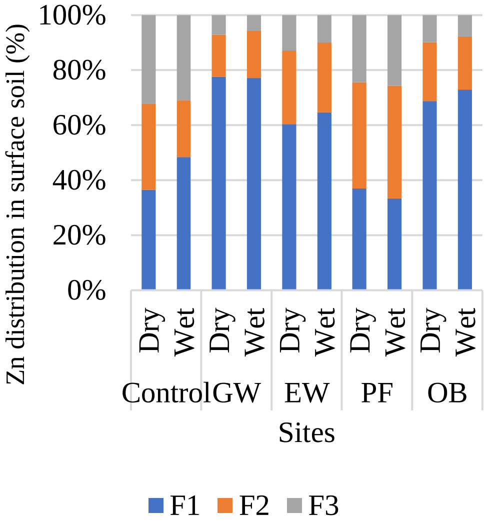


Figure 43 Distribution of Zn fractionation in surface soil

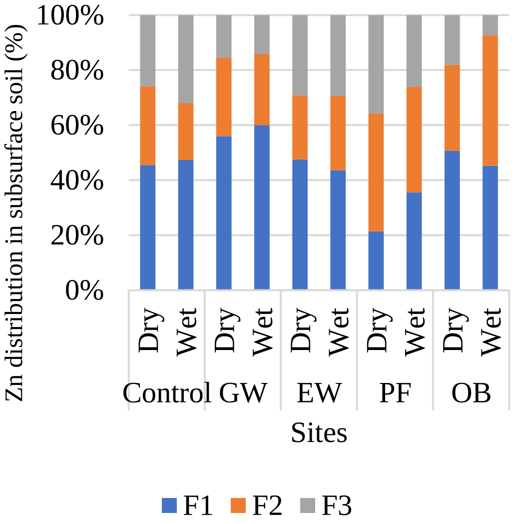


Figure 44 Distribution of Zn fractionation in subsurface soil

Figure 44 presents the percentage distribution of Zn fractionation in subsurface soil in the study area. The distribution of Zn in the subsurface soil was consistent with that in the surface soil. This can be concluded that F1 was the majority fraction in the control site, GW, EW, OB (only in dry season). For the Zn in the control site, F1 showed 45.5% in dry season and 47.3% in wet season; Zn in GW had F1 proportion for 56.0% in dry season and 60.1% in wet season. For Zn in EW, F1 accounted for 47.6% and 43.6% for dry and wet season. With the high contribution of F1, this indicated that their potential bioavailability was also high in the study area. In contrast, Zn in PF was mainly bound to F2 with a contribution of 42.8% in dry season, and 38.3% in wet season.

Statistical analysis of Zn chemical fraction showed that Zn in an exchangeable form (F1) was significantly higher than in F2, and F3 in control, EW, and GW. This indicated that Zn was significantly readily available and easily mobile in EW and GW environment. Zn could pose a potential risk to the organism in the area, especially plants grown in the household area of EW and GW. Besides, F1 and F2 of Zn were found significantly higher in OB than in the other sites, meaning that a significant amount of Zn in OB could easily mobile and potential bioavailable in the soil environment (Orroño & Lavado, 2009).

4.3 Risk assessment

The ecological risk posed by the heavy metal contaminated in the soils at the study area was assessed by the potential ecological risk index (PERI), based on the total concentration of the metals and their toxicity, and the risk assessment code (RAC), which utilized the potential mobility of the metals in the soils. The results of ecological risk assessment using PERI, and RAC were in 4.3.1 and 4.3.2, respectively. The PERI and RAC in each sampling site are shown in appendix F.

4.3.1 Potential ecological risk assessment

Based on the consideration of the total concentration of heavy metals and toxic-response factors of heavy metals, the potential ecological risk assessment was adopted to comprehensively evaluate the combined metal pollution risk threatening the ecosystem in the study area. The ecological risk factors (Er) of the studied metals in surface and subsurface soil were presented in Figure 45 and Figure 46, respectively.

Figure 45 showing Er for metals in surface soil, all metals, except for Cd, in the control site posed a low ecological risk with Er being lower than 40. At non-e-waste dismantling sites (NE), the studied heavy metal posed a low ecological risk (Er < 40). Similarly, all the heavy metals posed a low ecological risk to the groundwater well area (GW). Regarding the Er value of each metal in the e-waste dismantling houses (EW), Mn, Cr, Ni, Zn, As, and Pb also posed a low ecological risk. As shown in Figure 45, Cd posed a moderate ecological risk to 20% of the total soil samples collected from the EW. In addition, 60% of the samples collected from the EW showed low ecological risk, and 40% of the samples showed moderate ecological risk posed by Cu.

Considering the Er of paddy fields (PF), Mn, Cr, Ni, Zn, As, and Pb showed a low ecological risk level for all samples. Whilst, Cd posed ecological risk ranging from low to considerable levels, 50%, 34 %, and 16% of total soil samples posed low, moderate, and considerable ecological risk, respectively. For Cu, low and moderate ecological risk accounted for 84% and 16% of total soil samples.

At the open-burning site (OB), Mn, Cr, Ni, Zn, and As gave the low ecological risk to all soil samples. Pb posed low, moderate, and considerable ecological risk, which contributed to 50%, 25%, and 25% of total soil samples, respectively. Likewise, three risk levels consisting of low (25%), moderate (25%), and considerable (50%) were assessed from Cd contamination. Moreover, 25% and 75% of the soil samples contaminated Cu resulted in moderate and extreme ecological risk at OB, respectively.

With respect to the Er determined by individual metals in accordance with the sampling sites, the results reveal that of all the studied metals, Cd and Cu posed higher ecological risk at EW (moderate level), and PF area (moderate and considerable). Additionally, among all the studied elements, Pb, Cd, and Cu posed higher levels of ecological risk level at OB (moderate, considerable, and extreme). The results suggested that soil samples collected from EW and PF were more polluted with Cd and Cu compared to the other sites, while contamination of Pb, Cd, and Cu in the soils at OB were higher than those at the other sites.

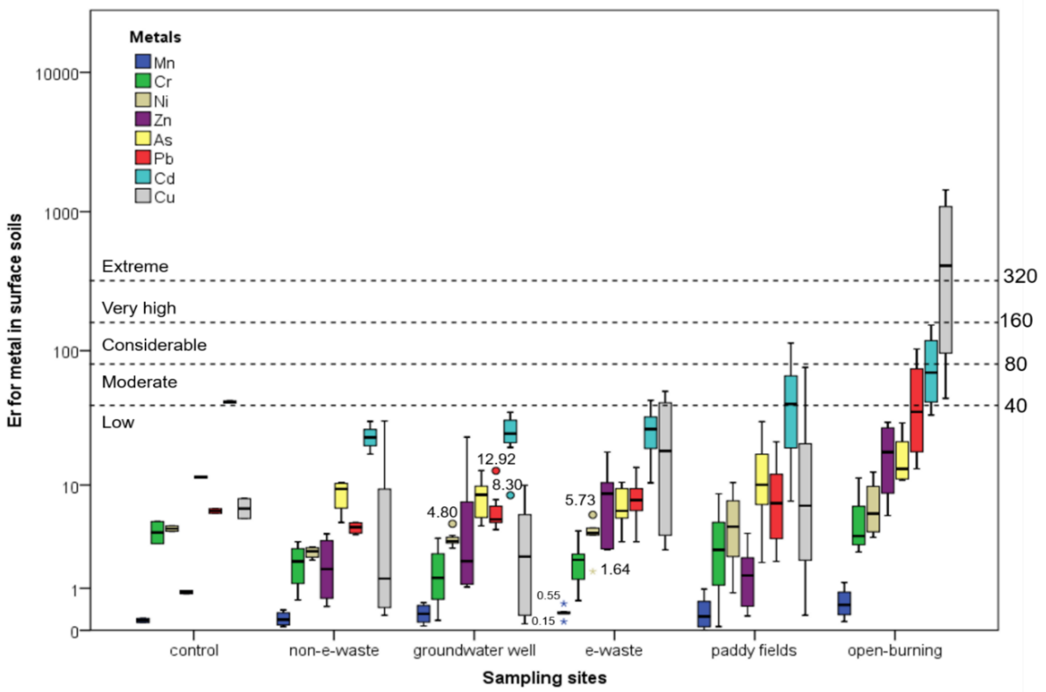


Figure 45 Ecological risk evaluated from the metals contamination in surface soils

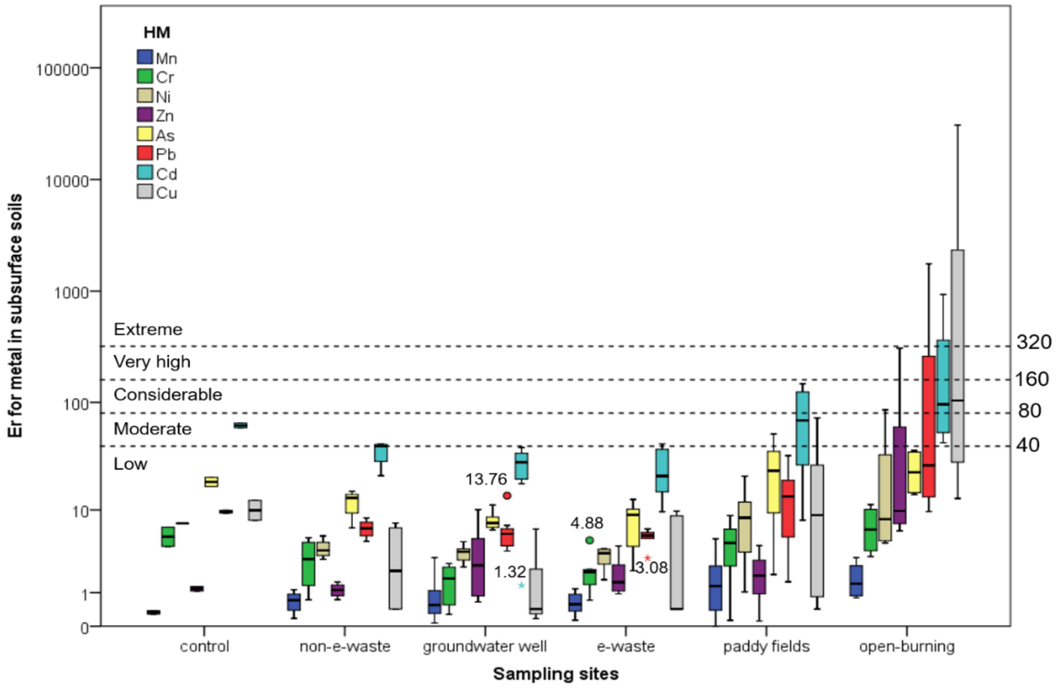


Figure 46 Ecological risk evaluated from the metals contamination in subsurface soils

The result could be supported by the study of Han et al. (2019), reporting that dismantling and burning of printed circuit board wastes lead to serious Cu, Cd, and Pb pollution in soil. In addition, the higher risk level of Cu, Pb, and Cd also occurred in paddy fields near the burning sites in Longtan, China (Q. Wu et al., 2015). Also, Cd posed a high risk in the surface soils of Qingyuan and moderate ecological risk in Guiyu, China, where burning of e-waste and dumping of e-waste after-burned residues were done in the area (He et al., 2017). Thus, the results in this study and other studies revealed that primitive e-waste dismantling and recycling, such as manual dismantling and burning of e-waste, could lead to Cu, Pb, and Cd pollution in surface soil at the open-burning site, and the surrounding area.

The result of Er for all metals in subsurface soil is illustrated in Figure 46, the Er at control, NE, and GW showed the low level of ecological risk as same as in the surface soils. Similarly, the Er of all metals in subsurface soil at EW showed a low ecological risk level. Unlike that of control, NE, GW, and EW, Er of As, Cd, and Cu posed a higher ecological risk in PF; As and Cu posed ecological risk at the PF with the range of low to moderate level. In contrast, Cd posed low to considerable ecological risk level in subsurface PF.

The Er of multiple metals posed ecological risk level of being higher than low level; Ni showed the ecological risk ranging from low to considerable level. Pb posed ecological risk levels that ranged from low to extremely high. Also, Cd was classified as moderate to extreme level of ecological risk. Furthermore, Cu was reflected board range from low (Er = 13) to extreme (Er = 30,000) ecological risk, which was 1.5 to 3,750 times higher than Er in the control sites. The relatively high value of Er was found in the subsurface layer in OB because the new layer of surface soils was refilled, so the previous surface soils have become the subsurface soil at OB. Therefore, the subsurface layers contained higher concentration of metals than the new surface layers; consequently, the higher metal concentration led to higher Er in the OB subsurface.

Figure 47 and Figure 48 show the potential ecological risk index (PERI) that expressed the overall risk posed by all studied metals to the surface and subsurface soil, respectively. Regarding Figure 47, PERI at the OB site was in the range of low to very high potential ecological risk, mainly contributed to Cu closely followed by Cd and Pb > Zn > As > Ni > Cr > Mn. PERI at the PF posed low to moderate ecological risk, primarily contributed to Cd > Cu > As > Pb > Ni > Cr > Zn > Mn. Conversely, those of low ecological risk were evaluated at control, NE, GW, and EW.

Regarding the PERI in subsurface soil shown in Figure 48, PERI in subsurface soil posed the same range of ecological risk as in surface soil. PERI posed low to very high risk to the ecosystem at the OB and low to moderate at the PF. On the other hand, PERI of control, EW, NE, and GW sites were elevated as low ecological risk. The most severe level of ecological risk (very high risk) was presented in the open-burning site in both surface and subsurface soil. This might be a consequence of uncontrolled e-waste burning activities and dumping unwanted electronic materials in this area.

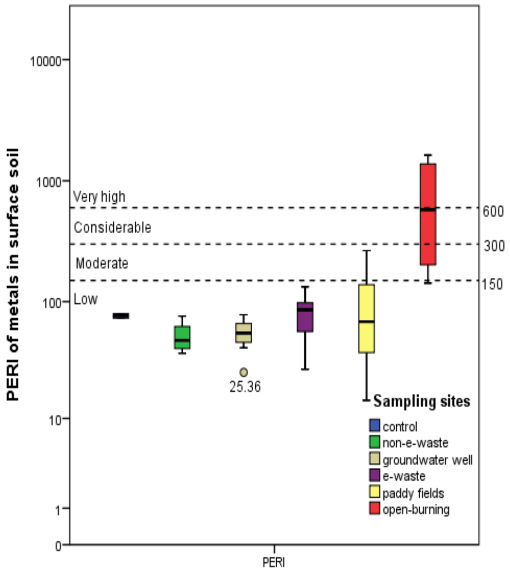


Figure 47 Potential ecological risk index (PERI) of metals in surface soil

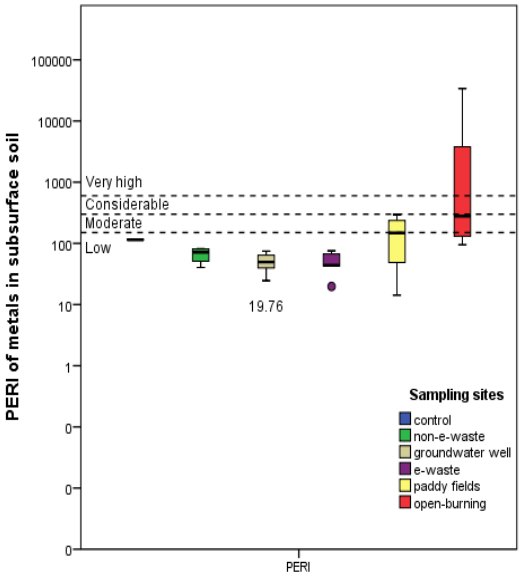


Figure 48 Potential ecological risk index (PERI) of metals in subsurface soil

Figure 49 and Figure 50 present the contribution of Er for the studied metal in surface and subsurface soil, respectively, which gave the same illustration. Er of Cd was the main contributor to PERI in all sampling sites, except for the OB, that of Cu showed the most significant contribution. These could result from the high value of the toxic-response factor of Cd, together with the Cu concentration that found the highest among all studied metals at the open-burning site. Similarly to Han et al. (2019) and Gullett et al. (2007), they reported that the open-burning of e-waste had emitted high amounts of heavy metals to the environment, so the open-burning at the e-waste site could lead to the high ecological risk caused by the heavy metal pollution. As the highest ecological risk posed by all the studied metals found at the open-burning site, the e-waste burning and dumping activities carried out in the site should be of great concern for the highest ecological risk from the pollution caused by the studied metals.

To identify the more explicit spatial contribution of the Er and PERI throughout the e-waste dismantling community, risk mapping of the Er and PERI were created using ArchMap with the method of Inverse Distance Weighted (IDW) technique. The risk maps of Er and PERI derived from the metals contamination in surface and subsurface soil were illustrated in Figure 51 and Figure 52, respectively.

Both figures presented a similar distribution pattern of Er in all metals of interest. The spatial distribution of Er in all the metals showed that the highest values of Er were found in the open-burning sites (OB) for both surface and subsurface soil. In addition, the highest Er of Zn was also found in surface soil at GW (code GW-2) that was located in the household area of one e-waste dismantling house. Likewise, the PERI, the combination of Er of the metals, had the highest value at the OB as well. With the highest values of Er and PERI apparently distributed in OB and GW (only Er of Zn), this suggested that OB could be considered as the hot spot of the ecological risks posed by all the heavy metals, and GW could also be considered the hot spot of Zn in surrounding area of the e-waste dismantling house. Consequently, it could be concluded that the contaminations of the studied elements were directly influenced by the e-waste dismantling and burning activities in the study area.

Apart from the highest Er and PERI found in the OB, PF had a secondly-high risk level in the study area. The results indicated that the ecological risk in the PF could be influenced by the e-waste burning activities carried out at the OB. The burning of e-waste in the open-air environment was reported to cause the pollution of heavy metals in both open-burning sites, and surrounding areas. The surrounding paddy fields, located 50-200 m away from the dismantling and burning of e-waste locations, were lower contaminated by the heavy metal generated from the burning activities through wet and dry deposition (Jinhui et al., 2011; Jun-hui & Hang, 2009). Regarding the Er and PERI found in the OB and PF, the metal pollution in the OB and PF could pose hazards to the local community.

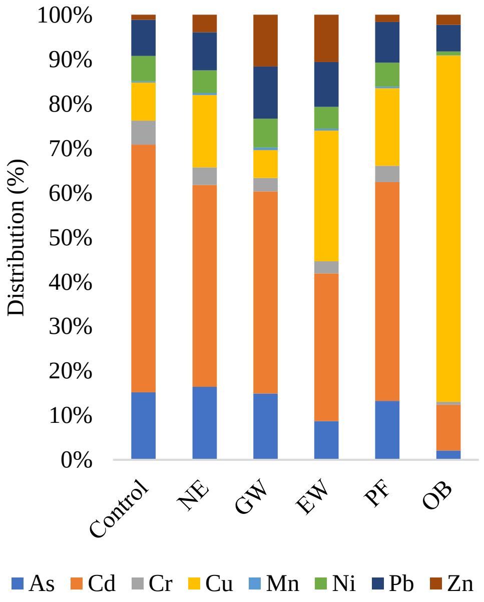


Figure 49 Er contributions to PERI in surface soil

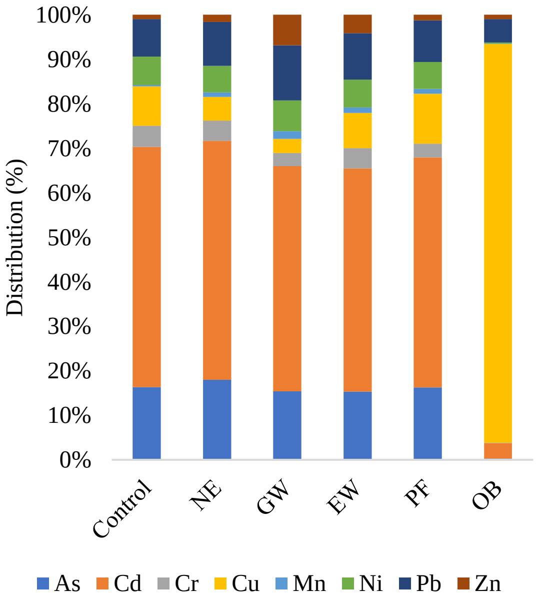


Figure 50 Er contributions to PERI in subsurface soil

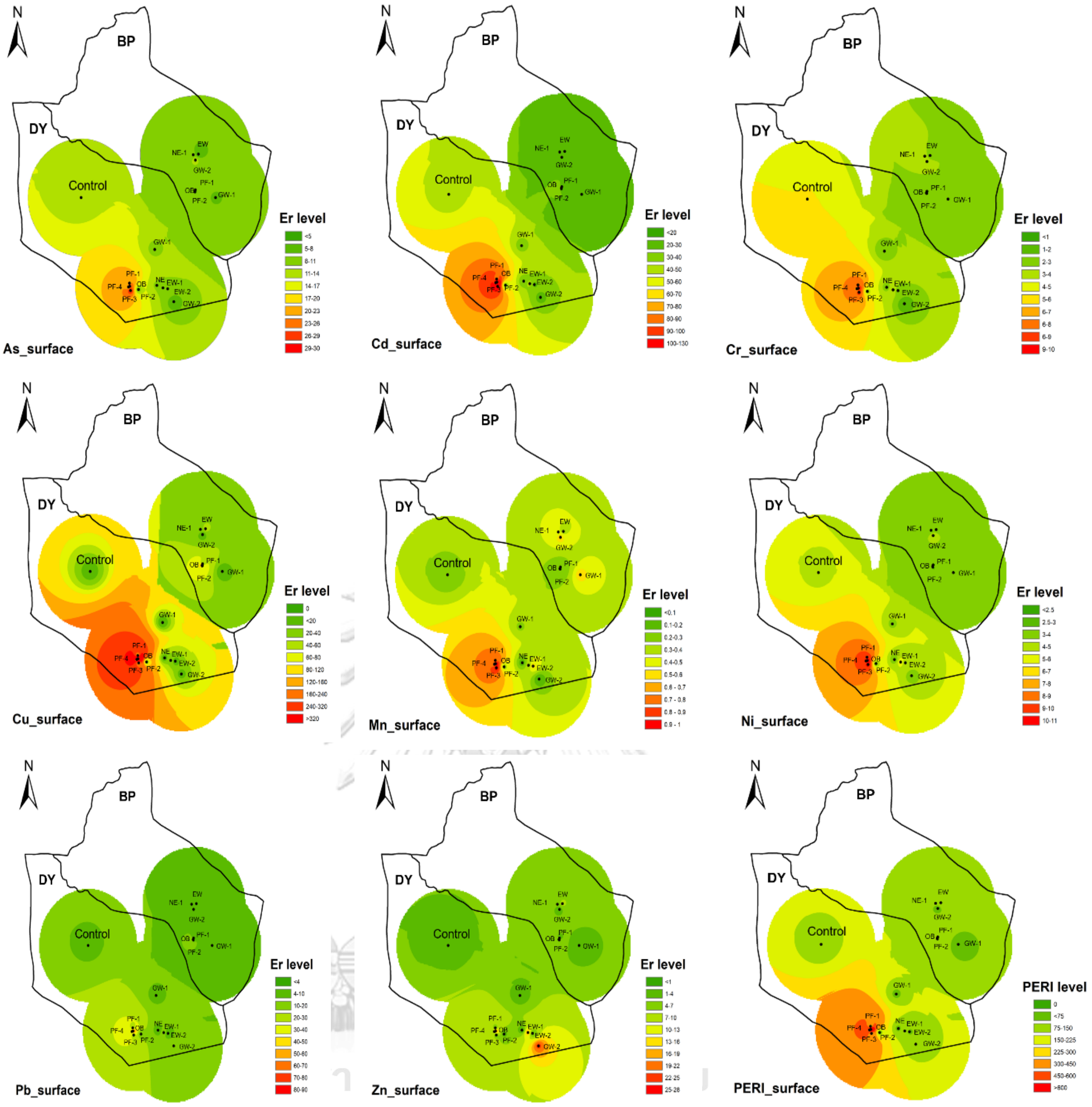


Figure 51 Distribution maps of Er and PERI in surface soil in the study area; DY, and BP refer to Daeng Yai subdistrict, and Ban Pao subdistrict, respectively.

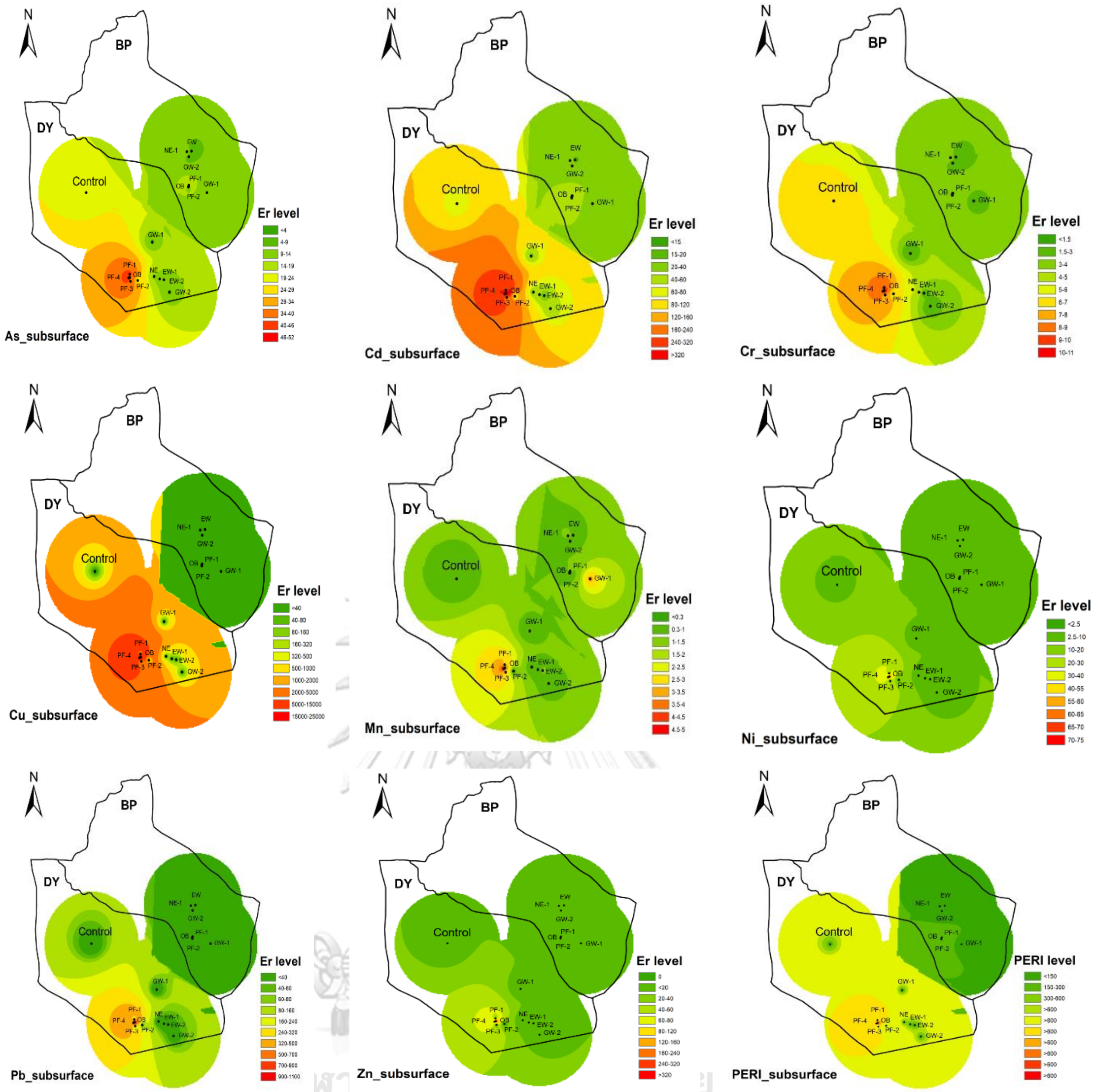


Figure 52 Distribution maps of Er and PERI in subsurface soil in the study area; DY, and BP refer to Daeng Yai subdistrict, and Ban Pao subdistrict, respectively.

4.3.2 Risk assessment code (RAC)

The RAC was applied to determine the environmental risk of heavy metal in the study area. The RAC was calculated based on the percentage of the exchangeable and weak acid soluble fractions (F1) that are highly bioavailable and readily mobile in the environment leading to environmental concerns. The results obtained from the RAC are classified into five levels and showed in Figure 53 and Table 20.

Regarding Figure 53, based on the F1 percentage in the control site, the studied metals showed a wide range of risk levels depending on the portion of each metal presenting in F1. Pb, Cd, Ni, As, and Cu completely showed no risk to low risk with the F1 being lower than 10%, while Cr, Mn, and Zn in some samples collected from the control site showed the ranging of risk between low to a high level. However, the median of the studied heavy metals shown in Table 20 revealed that Cd and Pb presented no risk to the control site, whereas As, Cr, Cu, and Mn posed low risk to the control site. Additionally, Mn (with the median F1 of 27.3%) and Zn (with the median F1 of 38.2%) posed medium, and high risk to the control site, respectively.

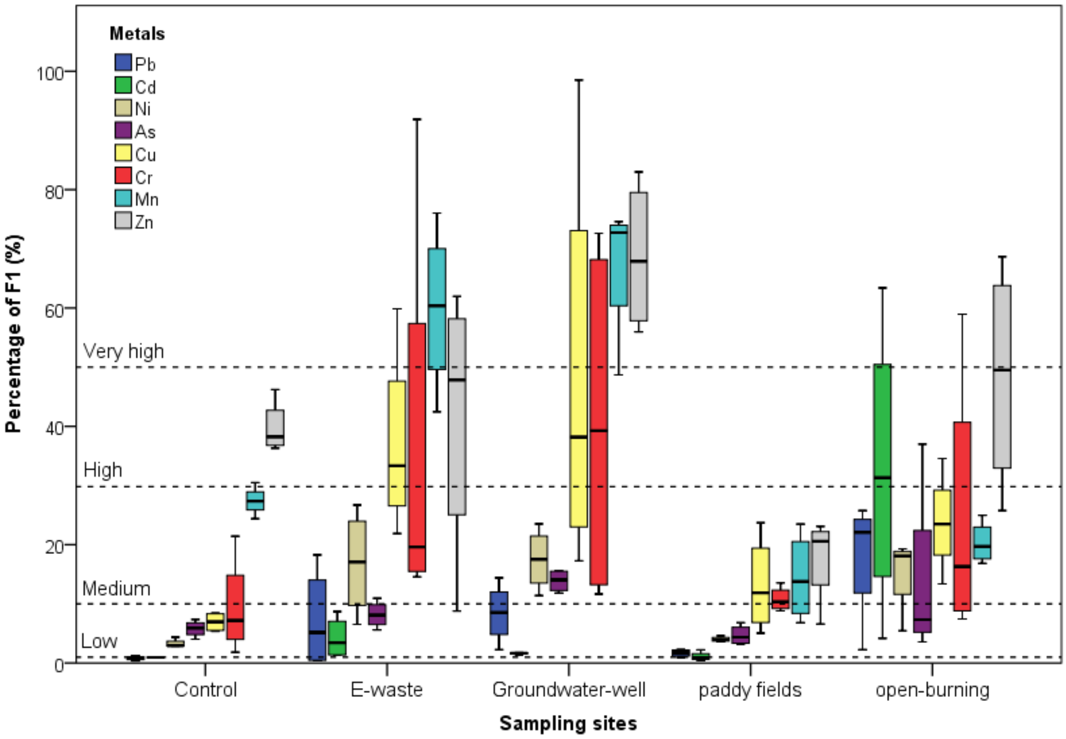


Figure 53 Risk assessment code (RAC) for the heavy metals in the study area

Regarding the median percentage heavy metal in soil bound to F1 in e-waste dismantling houses (EW), heavy metals in F1 were showed in the order of Mn (60.38) > Zn (47.85) > Cu (33.32) > Cr (19.62) > Ni (17.07) > As (8.09) > Pb (5.15) > Cd (3.44). This means that Mn had the highest percentage of F1, so Mn could pose the highest risk for the mobility in soils environment in the EW.

Table 20 Heavy metals posing a risk based on RAC

Risk level	**Percentage of F1**	**Sample sites**				
		Control	GW	EW	OB	PF
No risk	<1	Cd, Pb	-	-	-	Cd
Low risk	1-10	As, Cr, Cu, Ni	Cd, Pb	As, Cd, Pb	As	As, Cr, Ni, Pb,
Medium risk	11-30	Mn	As, Ni	Cr, Ni	Cr, Cu, Mn, Ni, Pb	Cu, Mn, Zn
High risk	31-50	Zn	Cr, Cu,	Cu, Mn, Zn	Cd	-
Very high risk	>50	-	Mn, Zn	-	Zn	-

Remark: The median of F1 of individual metals was used to represent the risk level posed by the metal.

In groundwater-well area (GW), the studied metals bound to F1 posed broad range risk between low and very high risk level (Figure 53), but the median of F1 in each metal showed that Cd and Pb were classified as low risk level, while As and Ni posed a medium risk. Cr and Cu posed a high risk, and Mn and Zn posed very high risk with the median F1 of 72.72% and 67.89%, respectively.

With respect to the percentage F1 on heavy metal in paddy fields (PF), all the studied heavy metals were reported to pose a risk between no risk and medium risk level with the median of F1 being lower than 30%. Cd was in the level of no risk (0.84%), while As, Cr, Ni, and Pb were in the level of low risk with the F1 ranging between 1% and 10%; Cu, Mn, and Zn were in the level of medium risk with the median F1 of 11.86%, 13.76%, and 20.57%, respectively. The result suggested that of all metals in PF, Cu, Mn, and Zn were more readily bioavailable in the paddy soils; Cu, Mn, and Zn partitioned to F1 had the highest bioavailability to rice grown in the paddy soils; they were considerable to take up by rice easily, and eventually, enter food chains. The results are supported by Zhao et al. (2011), revealing that the exchangeable phase (F1) of metals is the first fraction being easily solute and considered readily available to plants. However, Mn, Zn, and Cu are essential microelements for rice and other plants. Rice is one of the most Mn, Zn, and Cu-tolerant plants (Chen et al., 2013). Mn, Zn, and Cu were found to be accumulated highest in rice roots and rice leaves compared to in rice grains (Cui et al., 2019; Liu et al., 2017; Somayanda et al., 2013). The results from the previous studies indicated that Mn, Zn, and Cu showed low allocation to rice grains, so Mn, Zn, and Cu might not pose a potential ecological risk to the paddy fields.

Lastly, according to F1 percentage of the metals in open-burning sites (OB), the studied metals posed a wide range of risks from low to very high risk. As was in the level of low risk with the median F1 of 7.32%, while Cr (16.31%), Cu (23.49%), Mn (19.68%), Ni (18.08%), and Pb (22.08%) were in the level of medium risk. Moreover, Cd (31.31%) and Zn (49.53%) posed a high risk, and very high risk, respectively.

Overall, Zn and Mn showed the highest level of risk on the basis of potential mobility and bioavailability of heavy metal occurring in all soil sampling sites in the study area. In particular, Zn with very high risk level at the open-burning sites, and Zn, Cu, and Mn posing medium risk in the paddy fields, should be highly concerned because of the bioavailability and toxicity of the metals potentially causing severe soil contaminations and human impacts. Additionally, further plans related to agricultural purposes in the surrounding area of the open-burning sites should carefully considered due to the potential mobility and toxicity of the mentioned elements.

With respect to two methods of heavy metal risk assessment in soils at the study site (PERI and RAC method), this study revealed various risk levels caused by the studied heavy metals to the ecosystem in the study site. According to PERI method, Cd had the largest contribution to potential ecological risk values in the study sites on the basis of total heavy metal concentrations in the soils and toxicity-response factors of the metals, while Cu posed highest risk level to the ecosystem at the OB sites due to its highest content in the OB. RAC method, focusing on metal speciation and metal mobility, showed that Mn and Zn posed the highest risk level in all sampling sites because their highest percentage of F1 that is an index fraction representing the mobility and bioavailability of metals in the environment. Even though these two methods of ecological risk assessment posed by the heavy metals did not show consistent results, the methods represent the comprehensive risk assessment showing the ecological effects as a result of the heavy metal contamination in soils at the study area. However, in the OB, risk level assessed by the PERI method showed the highest PERI value among all sampling sites, and RAC also revealed that the highest risk level (very high risk) was found at the OB. This similarity indicated that the OB had the highest risk caused by the heavy metal contamination in soil resulted from the open-burning and open-dumping of e-waste and unwanted materials in the site.

The comparative results of PERI and RAC were presented in Table 21. The PERI (based on the heavy metal contents and toxicity) and RAC (based on the mobility of heavy metals) did not show the same results of ecological risk. To comprehensively sum up the ecological risk in the study site, the toxicity and the mobility of heavy metals were taken into consideration. After consideration of both toxicity and mobility of heavy metal, the result revealed that Cu potentially posed a risk to EW, and PF, and Cd, Cu, Pb, Ni, and Zn potentially posed a risk to OB.

Table 21 Comparative results of PERI and RAC method

Site	PERI		RAC (Risk levels)					Heavy metal potentially posing risk
	Surface	Subsurface	No	Low	Medium	High	Very high	
Control	Moderate: Cd		Cd, Pb	As, Cr, Cu, Ni	Mn	Zn	-	
NE	Low: all metals	Low: all metals			-			
GW			-	Cd, Pb	As, Ni	Cr, Cu	Mn, Zn	
EW	Moderate: Cd, Cu		-	As, Cd, Pb	Cr, Ni	Cu, Mn, Zn	-	Cu
PF	Moderate: Cu Considerable: Cd	Moderate: As, Cu Considerable :Cd	Cd	As, Cr, Ni, Pb	Cu, Mn, Zn	-	-	Cu
OB	Considerable: Pb, Cd Extreme: Cu	Considerable: Ni Very high :Zn Extreme: Pb, Cu, Cd	-	As	Cr, Cu, Mn, Ni, Pb	Cd	Zn	Cd, Cu, Pb, Ni, Zn

Chapter 5

Conclusions and Recommendations

5.1 Conclusions

The main objective of this study was to evaluate the potential risk posed by heavy metals in soils at the e-waste dismantling site and surrounding areas. There were three sub-objectives in the study. (1) To investigate the total contamination of heavy metals in soils at the e-waste site and surrounding area, (2) To investigate the fractions of the heavy metals in the e-waste site and surrounding areas using a sequential extraction approach, and (3) To evaluate the ecological risk posed by heavy metals in the e-waste site and surrounding areas. The soil samples were collected from the e-waste sites (e-waste dismantling houses, and e-waste open-burning site), and the surrounding areas (paddy fields, non-e-waste dismantling houses, and groundwater-well areas). The total concentration and the concentration of fractionation of As, Cd, Cr, Cu, Mn, Ni, Pb, and Zn were analyzed. The results of the study are concluded as follows:

1. E-waste dismantling activities operated at the open-burning sites and e-waste dismantling houses have caused the heavy metal contamination in soils at the e-waste sites and surrounding area.
2. The e-waste activities carried out in the e-waste dismantling houses and the open-burning sites have resulted in higher enrichment of Cu, Zn, and Pb compared to the sites where e-waste activities are not operated.
3. The paddy fields as a surrounding area of the e-waste sites were contaminated with heavy metals as a result of heavy metal released from the e-waste burning activities in the open-burning site.
4. Regarding the intervention values and Thai standard for residential and agricultural soils, As, Cu, Pb, and Zn in the open-burning sites showed serious contamination. Consequently, the open-burning sites needed some site remediations.
5. According to the mobility and bioavailability of heavy metal, Mn and Zn were found to have the highest potential mobility in the site, which led to a potential risk to soil ecosystem in the study area.
6. Based on the heavy metal contents, toxicity and bioavailability, e-waste activities in the open-burning site and e-waste dismantling houses have posed potential ecological risk caused by Cd, Cu, Pb, Ni, and Zn. In addition, Cu posed a potential ecological risk to the paddy fields, the surrounding sites of the open-burning sites.

5.2 Recommendations

The recommendations in the study were for two main groups of people evolved in the e-waste dismantling activities in the study area. The first group is local administrative organizations, and the second group is the e-waste workers.

Local administrative organization

1. An effective remediation technique for HM contamination on the OB is necessary to prevent the dissemination of HM to the surrounding area.
2. A local e-waste storage area and e-waste separating workshop should be built to serve formal e-waste dismantling activities and dumping of e-waste.
3. The local organization should promote best practices of e-waste dismantling activities to e-waste workers like indoor storage and concrete ground in the household e-waste dismantling.
4. Land use planning/zoning need to be implemented in order to separate agricultural area and e-waste dismantling/burning area.

E-waste workers

1. The workers should be encouraged to stop burning e-waste (wire cables) and use cable wire stripping machines together with personal protective equipment.
2. The household e-waste dismantling activities and piling of e-waste are recommended to operate above supporting liners to prevent the direct dispersion of HM-containing scarps/particulates to soils.
3. E-waste separating areas should be separated from the residential area, and vegetable garden in the household.

5.3 Limitation of the study

As heavy metals might occur in more than one oxidation state, which can present different degrees of the toxicity, the study of heavy metal oxidation state should be conducted for further study to explain precisely the toxicity of heavy metals in the study area.

REFERENCES

Ackah, M. (2017). Informal E-waste recycling in developing countries: review of metal (loid) s pollution, environmental impacts and transport pathways. *Environmental Science and Pollution Research, 24*(31), 24092-24101.

Addis, W., & Abebaw, A. (2017). Determination of heavy metal concentration in soils used for cultivation of Allium sativum L. (garlic) in East Gojjam Zone, Amhara Region, Ethiopia. *Cogent Chemistry, 3*(1). doi:10.1080/23312009.2017.1419422

Adelekan, B., & Alawode, A. (2011). Contributions of municipal refuse dumps to heavy metals concentrations in soil profile and groundwater in Ibadan Nigeria. *Journal of Applied Biosciences, 40*, 2727-2737.

Ahmadipour, F., Bahramifar, N., & Mahmood Ghasempouri, S. (2014). Fractionation and mobility of cadmium and lead in soils of Amol area in Iran, using the modified BCR sequential extraction method. *Chemical Speciation & Bioavailability, 26*(1), 31-36.

Alloway, B. J. (2012). *Heavy metals in soils: trace metals and metalloids in soils and their bioavailability* (Vol. 22): Springer Science & Business Media.

Amphalop, N., Suwanarat, N., & Prueksasit, T. (2017). *Heavy metal concentration in soil in e-waste separating household area, Buriram Province, Thailand*. Senior project. Department of Environmetal Science, Faculty of Science. Chulalongkorn University. Bangkok.

Amphalop, N., Suwantarat, N., Prueksasit, T., Yachusri, C., & Srithongouthai, S. (2020). Ecological risk assessment of arsenic, cadmium, copper, and lead contamination in soil in e-waste separating household area, Buriram province, Thailand. *Environmental Science and Pollution Research*, 1-16.

Andersson, A. (1979). On the distribution of heavy metals as compared to some other elements between grain size fractions in soils. *Swedish Journal of Agricultural Research*.

Ashraf, M. A., Maah, M. J., & Yusoff, I. (2012). Chemical Speciation and Potential Mobility of Heavy Metals in the Soil of Former Tin Mining Catchment. *The Scientific World Journal, 2012*, 11. doi:10.1100/2012/125608

Barbieri, M. (2016). The importance of enrichment factor (EF) and geoaccumulation index (Igeo) to evaluate the soil contamination. *J Geol Geophys, 5*(1), 1-4.

Borgese, L., Federici, S., Zacco, A., Gianoncelli, A., Rizzo, L., Smith, D. R., Donna, F., Lucchini, R., Depero, L. E., & Bontempi, E. (2013). Metal fractionation in soils and assessment of environmental contamination in Vallecamonica, Italy. *Environmental science and pollution research international, 20*(7), 5067-5075. doi:10.1007/s11356-013-1473-8

Bouyoucos, G. J. (1962). Hydrometer method improved for making particle size analyses of soils 1. *Agronomy Journal, 54*(5), 464-465.

Brigden, K., Labunska, I., Santillo, D., & Allsopp, M. (2005). Recycling of electronic wastes in China and India: workplace and environmental contamination. *Greenpeace International, 55*.

CEPA. (1995). Environmental Quality Standard for Soils (GB15618-1995). In: CEPA Beijing, China.

CEPA, C. E. P. A. (2007). Canadian soil quality guidelines for the Protection of Environmental and Human Health.

Chen, Z., Fujii, Y., Yamaji, N., Masuda, S., Takemoto, Y., Kamiya, T., Yusuyin, Y., Iwasaki, K., Kato, S.-i., Maeshima, M., Ma, J. F., & Ueno, D. (2013). Mn tolerance in rice is mediated by MTP8.1, a member of the cation diffusion facilitator family. *Journal of experimental botany, 64*(14), 4375-4387. doi:10.1093/jxb/ert243

Chuan, M., Shu, G., & Liu, J. (1996). Solubility of heavy metals in a contaminated soil: effects of redox potential and pH. *Water, Air, and Soil Pollution, 90*(3-4), 543-556.

Cui, J.-l., Zhao, Y.-p., Lu, Y.-J., Chan, T.-s., Zhang, L.-l., Tsang, D. C. W., & Li, X.-d. (2019). Distribution and speciation of copper in rice (Oryza sativa L.) from mining-impacted paddy soil: Implications for copper uptake mechanisms. *Environment International, 126*, 717-726. doi:https://doi.org/10.1016/j.envint.2019.02.045

Damrongsiri, S., Vassanadumrongdee, S., & Tanwattana, P. (2016). Heavy metal contamination characteristic of soil in WEEE (waste electrical and electronic equipment) dismantling community: a case study of Bangkok, Thailand. *Environmental Science and Pollution Research, 23*(17), 17026-17034. doi:https://doi.org/10.1007/s11356-016-6897-5

DEFRA, E. (2002). Soil guideline values for arsenic contamination. *London: Department for the Environment, Food and Rural Affairs and the Environment Agency. Bristol, England*.

Dudka, S., Ponce-Hernandez, R., Tate, G., & Hutchinson, T. (1996). Forms of Cu, Ni, and Zn in soils of Sudbury, Ontario and the metal concentrations in plants. *Water, Air, and Soil Pollution, 90*(3-4), 531-542.

EPA, U. (1996). Method 3050B: acid digestion of sediments, sludges, and soils. *Environmental Protection Agency: Washington, DC, USA*.

EPA, U. (2001). Supplemental guidance for developing soil screening levels for superfund sites. *Peer Review Draft, OSWER, 9355*, 4-24.

Essington, M. E. (2015). *Soil and water chemistry: an integrative approach*: CRC press.

Evans, L. (1989). Chemistry of metal retention by soils. *Environmental Science & Technology, 23*(9), 1046-1056.

Fairbrother, A., Wenstel, R., Sappington, K., & Wood, W. (2007). Framework for Metals Risk Assessment. *Ecotoxicology and Environmental Safety, 68*(2), 145-227. doi:https://doi.org/10.1016/j.ecoenv.2007.03.015

Favas, P. J. C., Pratas, J., Gomes, M. E. P., & Cala, V. (2011). Selective chemical extraction of heavy metals in tailings and soils contaminated by mining activity: Environmental implications. *Journal of Geochemical Exploration, 111*(3), 160-171. doi:https://doi.org/10.1016/j.gexplo.2011.04.009

Fiedler, S., Vepraskas, M. J., & Richardson, J. L. (2007). Soil Redox Potential: Importance, Field Measurements, and Observations. In D. L. Sparks (Ed.), *Advances in Agronomy* (Vol. 94, pp. 1-54): Academic Press.

Fosu-Mensah, B. Y., Addae, E., Yirenya-Tawiah, D., & Nyame, F. (2017). Heavy metals concentration and distribution in soils and vegetation at Korle Lagoon area in Accra, Ghana. *Cogent Environmental Science, 3*(1). doi:10.1080/23311843.2017.1405887

Fujimori, T., & Takigami, H. (2014). Pollution distribution of heavy metals in surface soil at an informal electronic-waste recycling site. *Environmental Geochemistry* *and Health, 36*(1), 159-168. doi:https://doi.org/10.1007/s10653-013-9526-y

Fujimori, T., Takigami, H., Agusa, T., Eguchi, A., Bekki, K., Yoshida, A., Terazono, A., & Ballesteros, F. C. (2012). Impact of metals in surface matrices from formal and informal electronic-waste recycling around Metro Manila, the Philippines, and intra-Asian comparison. *Journal of Hazardous Materials, 221-222,* 139-146. doi:https://doi.org/10.1016/j.jhazmat.2012.04.019

Gleason, S. M., Ewel, K. C., & Hue, N. (2003). Soil redox conditions and plant–soil relationships in a micronesian mangrove forest. *Estuarine, Coastal and Shelf Science, 56*(5), 1065-1074. doi:https://doi.org/10.1016/S0272-7714(02)00307-4

Gonnelli, C., & Renella, G. (2013). Chromium and nickel. In *Heavy metals in soils* (pp. 313-333): Springer.

Gullett, B. K., Linak, W. P., Touati, A., Wasson, S. J., Gatica, S., & King, C. J. (2007). Characterization of air emissions and residual ash from open burning of electronic wastes during simulated rudimentary recycling operations. *Journal of Material Cycles and Waste Management, 9*(1), 69-79. doi:10.1007/s10163-006-0161-x

Ha, N. N., Agusa, T., Ramu, K., Tu, N. P. C., Murata, S., Bulbule, K. A., Parthasaraty, P., Takahashi, S., Subramanian, A., & Tanabe, S. (2009). Contamination by trace elements at e-waste recycling sites in Bangalore, India. *Chemosphere, 76*(1), 9-15. doi:https://doi.org/10.1016/j.chemosphere.2009.02.056

Hakanson, L. (1980). An ecological risk index for aquatic pollution control. A sedimentological approach. *Water research, 14*(8), 975-1001.

Han, Y., Tang, Z., Sun, J., Xing, X., Zhang, M., & Cheng, J. (2019). Heavy metals in soil contaminated through e-waste processing activities in a recycling area: Implications for risk management. *Process Safety and Environmental Protection, 125,* 189-196. doi:https://doi.org/10.1016/j.psep.2019.03.020

He, K., Sun, Z., Hu, Y., Zeng, X., Yu, Z., & Cheng, H. (2017). Comparison of soil heavy metal pollution caused by e-waste recycling activities and traditional industrial operations. *Environmental Science and Pollution Research, 24*(10), 9387-9398. doi:https://doi.org/10.1007/s11356-017-8548-x

Isimekhai, K. A., Garelick, H., Watt, J., & Purchase, D. (2017). Heavy metals distribution and risk assessment in soil from an informal E-waste recycling site in Lagos State, Nigeria. *Environmental Science and Pollution Research, 24*(20), 17206-17219. doi:https://doi.org/10.1007/s11356-017-8877-9

Jinhui, L., Huabo, D., & Pixing, S. (2011). Heavy metal contamination of surface soil in electronic waste dismantling area: site investigation and source-apportionment analysis. *Waste Management & Research, 29*(7), 727-738. doi:https://doi.org/10.1177/0734242X10397580

Jun-hui, Z., & Hang, M. (2009). Eco-toxicity and metal contamination of paddy soil in an e-wastes recycling area. *Journal of Hazardous Materials, 165*(1), 744-750. doi:https://doi.org/10.1016/j.jhazmat.2008.10.056

Khalilova, H., & Mammadov, V. (2016). Assessing the Anthropogenic Impact on Heavy Metal Pollution of Soils and Sediments in Urban Areas of Azerbaijan's Oil Industrial Region. *Polish Journal of Environmental Studies, 25*(1).

Kong, X., Liu, T., Yu, Z., Chen, Z., Lei, D., Wang, Z., Zhang, H., Li, Q., & Zhang, S. (2018). Heavy Metal Bioaccumulation in Rice from a High Geological Background Area in Guizhou Province, China. *International journal of* *environmental research and public health, 15*(10), 2281. doi:10.3390/ijerph15102281

Kumar, A., Holuszko, M., & Espinosa, D. C. R. (2017). E-waste: An overview on generation, collection, legislation and recycling practices. *Resources, Conservation and Recycling, 122*, 32-42. doi:https://doi.org/10.1016/j.resconrec.2017.01.018

Kyere, V. N., Greve, K., & Atiemo, S. M. (2016). Spatial assessment of soil contamination by heavy metals from informal electronic waste recycling in Agbogbloshie, Ghana. *Environ Anal Health Toxicol, 31*(0), 2016006-2016000. doi:https://doi.org/10.5620/eht.e2016006

Lee, S. (2006). Geochemistry and partitioning of trace metals in paddy soils affected by metal mine tailings in Korea. *Geoderma, 135*, 26-37. doi:https://doi.org/10.1016/j.geoderma.2005.11.004

Leung, A. (2019). Chapter 15 - Environmental Contamination and Health Effects Due to E-waste Recycling. In M. N. V. Prasad & M. Vithanage (Eds.), *Electronic Waste Management and Treatment Technology* (pp. 335-362). Oxford: Butterworth-Heinemann.

Leung, A., Cai, Z. W., & Wong, M. H. (2006). Environmental contamination from electronic waste recycling at Guiyu, southeast China. *Journal of Material Cycles and Waste Management, 8*(1), 21-33. doi:https://doi.org/10.1007/s10163-005-0141-6

Li, W., & Achal, V. (2020). Environmental and health impacts due to e-waste disposal in China – A review. *Science of The Total Environment, 737*, 139745. doi:https://doi.org/10.1016/j.scitotenv.2020.139745

Liu, C., Chen, G., Li, Y., Peng, Y., Zhang, A., Hong, K., Jiang, H., Ruan, B., Zhang, B., Yang, S., Gao, Z., & Qian, Q. (2017). Characterization of a major QTL for manganese accumulation in rice grain. *Scientific reports, 7*(1), 17704-17704. doi:10.1038/s41598-017-18090-7

Lopez-Sanchez, J., Sahuquillo, A., Rauret, G., Lachica, M., Barahona, E., Gomez, A., Ure, A., Muntau, H., & Quevauviller, P. (2002). Extraction procedures for soil analysis. *Methodologies for soil and sediment fractionation studies*, 28-57.

Lopez, B. N., Man, Y. B., Zhao, Y. G., Zheng, J. S., Leung, A. O., Yao, J., & Wong, M. H. (2011). Major pollutants in soils of abandoned agricultural land contaminated by e-waste activities in Hong Kong. *Archives of Environmental Contamination and Toxicology, 61*(1), 101-114. doi:10.1007/s00244-010-9590-6

Luo, C., Liu, C., Wang, Y., Liu, X., Li, F., Zhang, G., & Li, X. (2011). Heavy metal contamination in soils and vegetables near an e-waste processing site, south China. *Journal of Hazardous Materials, 186*(1), 481-490. doi:https://doi.org/10.1016/j.jhazmat.2010.11.024

Manomaivibool, P., & Vassanadumrongdee, S. (2011). Extended Producer Responsibility in Thailand. *Journal of Industrial Ecology, 15*(2), 185-205. doi:10.1111/j.1530-9290.2011.00330.x

McCauley, A., Jones, C., & Jacobsen, J. (2009). Soil pH and organic matter. *Nutrient management module, 8*(2), 1-12.

Mclaren, R. G., & Crawford, D. V. (1973). STUDIES ON SOIL COPPER I. THE FRACTIONATION OF COPPER IN SOILS. *Journal of Soil Science, 24*(2), 172-181. doi:10.1111/j.1365-2389.1973.tb00753.x

MFE, M. f. t. E. (2011). *Methodology for Deriving Standards for Contaminants in Soil to Project Human Health*. Wellington: Ministry for the Environment.

Moeckel, C., Breivik, K., Nøst, T. H., Sankoh, A., Jones, K. C., & Sweetman, A. (2020). Soil pollution at a major West African E-waste recycling site: Contamination pathways and implications for potential mitigation strategies. *Environment International, 137*, 105563. doi:https://doi.org/10.1016/j.envint.2020.105563

Oguri, T., Suzuki, G., Matsukami, H., Uchida, N., Tue, N. M., Tuyen, L. H., Viet, P. H., Takahashi, S., Tanabe, S., & Takigami, H. (2018). Exposure assessment of heavy metals in an e-waste processing area in northern Vietnam. *Science of The Total Environment, 621*, 1115-1123. doi:https://doi.org/10.1016/j.scitotenv.2017.10.115

Olafisoye, O. B., Adefioye, T., & Osibote, O. A. (2013). Heavy metals contamination of water, soil, and plants around an electronic waste dumpsite. *Pol. J. Environ. Stud, 22*(5), 1431-1439.

Orroño, D. I., & Lavado, R. S. (2009). Distribution of extractable heavy metals in different soil fractions. *Chemical Speciation & Bioavailability, 21*(4), 193-198. doi:10.3184/095422909X12473204137916

Oyeyiola, A. O., Olayinka, K. O., & Alo, B. I. (2011). Comparison of three sequential extraction protocols for the fractionation of potentially toxic metals in coastal sediments. *Environmental Monitoring and Assessment, 172*(1), 319-327. doi:10.1007/s10661-010-1336-4

Pagano, G., Guida, M., Tommasi, F., & Oral, R. (2015). Health effects and toxicity mechanisms of rare earth elements—Knowledge gaps and research prospects. *Ecotoxicology and Environmental Safety, 115*, 40-48. doi:https://doi.org/10.1016/j.ecoenv.2015.01.030

Payus, C., & Talip, A. F. A. (2014). Assessment of heavy metals accumulation in paddy rice (Oryza sativa). *African Journal of Agricultural Research, 9*(41), 3082-3090.

Perin, G., Craboledda, L., Lucchese, M., Cirillo, R., Dotta, L., Zanette, M., & Orio, A. (1985). Heavy metal speciation in the sediments of northern Adriatic Sea. A new approach for environmental toxicity determination. *Heavy metals in the environment, 2*(1), 454-456.

Perkins, D. N., Brune Drisse, M.-N., Nxele, T., & Sly, P. D. (2014). E-Waste: A Global Hazard. *Annals of Global Health, 80*(4), 286-295. doi:https://doi.org/10.1016/j.aogh.2014.10.001

Poopa, T., Pavasant, P., Kanokkantapong, V., & Panyapinyopol, B. (2015). Fractionation and mobility of lead in Klity Creek riverbank sediments, Kanchanaburi, Thailand. *Applied Environmental Research, 37*(1), 1-10.

Pradhan, J. K., & Kumar, S. (2014). Informal e-waste recycling: environmental risk assessment of heavy metal contamination in Mandoli industrial area, Delhi, India. *Environmental Science and Pollution Research, 21*(13), 7913-7928. doi:10.1007/s11356-014-2713-2

Pueyo, M., Sastre, J., Hernández, E., Vidal, M., López-Sánchez, J. F., & Rauret, G. (2003). Prediction of trace element mobility in contaminated soils by sequential extraction. *Journal of Environmental Quality, 32*(6), 2054-2066. doi:10.2134/jeq2003.2054

Qian, J., Shan, X.-q., Wang, Z.-j., & Tu, Q. (1996). Distribution and plant availability of heavy metals in different particle-size fractions of soil. *Science of the Total Environment, 187*(2), 131-141. doi:https://doi.org/10.1016/0048-9697(96)05134-0

Quan, S. X., Yan, B., Yang, F., Li, N., Xiao, X. M., & Fu, J. M. (2015). Spatial distribution of heavy metal contamination in soils near a primitive e-waste recycling site. *Environmental Science and Pollution Research, 22*(2), 1290-1298. doi:10.1007/s11356-014-3420-8

Quevauviller, P., Rauret, G., Muntau, H., Ure, A., Rubio, R., López-Sánchez, J., Fiedler, H., & Griepink, B. (1994). Evaluation of a sequential extraction procedure for the determination of extractable trace metal contents in sediments. *Fresenius' journal of analytical chemistry, 349*(12), 808-814.

Rao, C. R. M., Sahuquillo, A., & Lopez Sanchez, J. F. (2008). A Review of the Different Methods Applied in Environmental Geochemistry For Single and Sequential Extraction of Trace Elements in Soils and Related Materials. *Water, Air, and Soil Pollution, 189*(1), 291-333. doi:10.1007/s11270-007-9564-0

Rauret, G., Lopez-Sanchez, J., Sahuquillo, A., Rubio, R., Davidson, C., Ure, A., & Quevauviller, P. (1999). Improvement of the BCR three step sequential extraction procedure prior to the certification of new sediment and soil reference materials. *Journal of Environmental Monitoring, 1*(1), 57-61.

Rieuwerts, J. S., Thornton, I., Farago, M. E., & Ashmore, M. R. (1998). Factors influencing metal bioavailability in soils: preliminary investigations for the development of a critical loads approach for metals. *Chemical Speciation & Bioavailability, 10*(2), 61-75. doi:https://doi.org/10.3184/095422998782775835

Sahuquillo, A., Lopez-Sanchez, J., Gleyzes, C., Ruban, V., Comans, R., Rauret, G., Ure, A., Gomez, A., Barahona, E., & Muntau, H. (2007). *Methodologies for soil and sediment fractionation studies*: Royal Society of Chemistry.

Sarkar, S. K., Favas, P. J., Rakshit, D., & Satpathy, K. (2014). Geochemical speciation and risk assessment of heavy metals in soils and sediments. In *Environmental risk assessment of soil contamination*: IntechOpen.

Scholz, M. (2019). Chapter 4 - Mineral and Biological Contamination of Soil and Crops Irrigated With Recycled Domestic Wastewater. In M. Scholz (Ed.), *Sustainable Water Treatment* (pp. 55-83): Elsevier.

Sims, J., & Patrick, W. (1978). The Distribution of Micronutrient Cations in Soil Under Conditions of Varying Redox Potential and pH 1. *Soil Science Society of America Journal, 42*(2), 258-262.

Somayanda, I., Morete, M. J., Ismail, A., Schulin, R., & Johnson-Beebout, S. (2013). Zn uptake, translocation and grain Zn loading in rice (Oryza sativa L.) genotypes selected for Zn deficiency tolerance and high grain Zn. *Journal of experimental botany, 64*. doi:10.1093/jxb/ert118

Sungur, A., Soylak, M., & Ozcan, H. (2014). Investigation of heavy metal mobility and availability by the BCR sequential extraction procedure: relationship between soil properties and heavy metals availability. *Chemical Speciation & Bioavailability, 26*(4), 219-230. doi:10.3184/095422914X14147781158674

Suzuki, G., Someya, M., Matsukami, H., Uchida-Noda, N., Tue, N., Fujimori, T., Tuyen, L., Agusa, T., Viet, P., & Takahashi, S. (2013). Emissions of Dioxin-like compounds from primitive E-waste recycling activities in the northern part of Vietnam− 1st report–. *Organohalogen Compounds, 75*, 1273-1277.

Tang, X., Shen, C., Shi, D., Cheema, S. A., Khan, M. I., Zhang, C., & Chen, Y. (2010). Heavy metal and persistent organic compound contamination in soil from Wenling: An emerging e-waste recycling city in Taizhou area, China. *Journal of Hazardous Materials, 173*(1), 653-660. doi:https://doi.org/10.1016/j.jhazmat.2009.08.134

Tessier, A., Campbell, P. G., & Bisson, M. (1979). Sequential extraction procedure for the speciation of particulate trace metals. *Analytical chemistry, 51*(7), 844-851.

Thai Office of National Environment Board. (2014). Standard level for residential soil. Retrieved from http://www.onep.go.th/topics/20748/

ThaiDMR. (2010). *Mineral resources management in Buriram, Thailand* (Vol. 1). Bangkok: Department of mineral resources of Thailand.

ThaiLDD. (2010). *Chemical Analysis of Soil Handbook* (Vol. 1). Bangkok: Land Development Department of Thailand.

ThaiPCD. (2019). *Pollution Situation in Thailand of 2018*. Bangkok: Ministry of Natural Resources and Environment.

Thongkaow, P., Prueksasit, T., & Siriwong, W. (2017, 8th-9th December). *Material flow of informal electronic waste dismantling in rural area of northeastern Thiland.* Paper presented at the 139th The IIER International Conference Osaka, Japan.

UNU. (2014). Solving the E-Waste Problem (Step) White Paper: One Global Definition of E-waste. *Bonn, Germany. Step, 3576*.

Vassanadumrongdee, S. (2014). Heavy metal contamination in soil in e-waste recycling area in Thailand. Retrieved from http://www.eric.chula.ac.th/download/ew58/ew_pocd.pdf

VROM. (2000). The Circular on Target Values and Intervention Values for Soil Remediation. *Ministry of Housing, Spatial Planning and the Environment, the Netherlands*.

Walkley, A., & Black, I. A. (1934). An examination of the Degtjareff method for determining soil organic matter, and a proposed modification of the chromic acid titration method. *Soil science, 37*(1), 29-38.

Wang, J., Liu, G., Lu, l., & Liu, H. (2015). Metal distribution and bioavailability in surface sediments from the Huaihe River, Anhui, China. *Environmental Monitoring and Assessment, 188*(1), 3. doi:10.1007/s10661-015-5005-5

Wedepohl, K. H. (1995). The composition of the continental crust. *Geochimica et Cosmochimica Acta, 59*(7), 1217-1232. doi:https://doi.org/10.1016/0016-7037(95)00038-2

Withayaanumas, S. (2017). E-waste management in Thailand. Retrieved from https://tdri.or.th/wp-content/uploads/2018/04/wb133.pdf

Wong, M. H., Wu, S. C., Deng, W. J., Yu, X. Z., Luo, Q., Leung, A. O., Wong, C. S., Luksemburg, W. J., & Wong, A. S. (2007). Export of toxic chemicals - a review of the case of uncontrolled electronic-waste recycling. *Environmental Pollution, 149*(2), 131-140. doi:10.1016/j.envpol.2007.01.044

Wu, Q., Leung, J. Y. S., Geng, X., Chen, S., Huang, X., Li, H., Huang, Z., Zhu, L., Chen, J., & Lu, Y. (2015). Heavy metal contamination of soil and water in the vicinity of an abandoned e-waste recycling site: Implications for dissemination of heavy metals. *Science of The Total Environment, 506-507*, 217-225. doi:https://doi.org/10.1016/j.scitotenv.2014.10.121

Wu, W., Wu, P., Yang, F., Sun, D.-l., Zhang, D.-X., & Zhou, Y.-K. (2018). Assessment of heavy metal pollution and human health risks in urban soils around an electronics manufacturing facility. *Science of The Total Environment, 630*, 53-61. doi:https://doi.org/10.1016/j.scitotenv.2018.02.183

Xian, X., & Shokohifard, G. I. (1989). Effect of pH on chemical forms and plant availability of cadmium, zinc, and lead in polluted soils. *Water, Air, and Soil Pollution, 45*(3-4), 265-273.

Xiong, L.-M., & Lu, R.-K. (1993). Effect of liming on plant accumulation of cadmium under upland or flooded conditions. *Environmental pollution, 79*(2), 199-203.

Xu, F., Liu, Y., Wang, J., Zhang, G., Zhang, W., Liu, L., Wang, J., Pan, B., & Lin, K. (2015). Characterization of heavy metals and brominated flame retardants in the indoor and outdoor dust of e-waste workshops: implication for on-site human exposure. *Environmental Science and Pollution Research, 22*(7), 5469-5480. doi:10.1007/s11356-014-3773-z

Yolcubal, I., Brusseau, M. L., Artiola, J. F., Wierenga, P., & Wilson, L. G. (2004). 12 - ENVIRONMENTAL PHYSICAL PROPERTIES AND PROCESSES. In J. F. Artiola, I. L. Pepper, & M. L. Brusseau (Eds.), *Environmental Monitoring and Characterization* (pp. 207-239). Burlington: Academic Press.

Yu, G. B., Liu, Y., Yu, S., Wu, S. C., Leung, A. O. W., Luo, X. S., Xu, B., Li, H. B., & Wong, M. H. (2011). Inconsistency and comprehensiveness of risk assessments for heavy metals in urban surface sediments. *Chemosphere, 85*(6), 1080-1087. doi:https://doi.org/10.1016/j.chemosphere.2011.07.039

Yuan, G.-L., Sun, T.-H., Han, P., Li, J., & Lang, X.-X. (2014). Source identification and ecological risk assessment of heavy metals in topsoil using environmental geochemical mapping: Typical urban renewal area in Beijing, China. *Journal of Geochemical Exploration, 136*, 40-47. doi:https://doi.org/10.1016/j.gexplo.2013.10.002

Zarcinas, B. A., Pongsakul, P., McLaughlin, M. J., & Cozens, G. (2004). Heavy metals in soils and crops in Southeast Asia 2. Thailand. *Environmental Geochemistry and Health, 26*(3), 359-371. doi:10.1007/s10653-005-4670-7

Zeng, F., Ali, S., Zhang, H., Ouyang, Y., Qiu, B., Wu, F., & Zhang, G. (2011). The influence of pH and organic matter content in paddy soil on heavy metal availability and their uptake by rice plants. *Environmental pollution, 159*(1), 84-91. doi:https://doi.org/10.1016/j.envpol.2010.09.019

Zhao, K., fu, W., Ye, Z., & Zhang, C. (2015). Contamination and Spatial Variation of Heavy Metals in the Soil-Rice System in Nanxun County, Southeastern China. *International journal of environmental research and public health, 12*, 1577-1594. doi:10.3390/ijerph120201577

Zhao, K., Liu, X., Zhang, W., Xu, J., & Wang, F. (2011). Spatial dependence and bioavailability of metal fractions in paddy fields on metal concentrations in rice grain at a regional scale. *Journal of Soils and Sediments, 11*(7), 1165. doi:10.1007/s11368-011-0408-6

Zhao, W., Ding, L., Gu, X., Luo, J., Liu, Y., Guo, L., Shi, Y., Huang, T., & Cheng, S. (2015). Levels and ecological risk assessment of metals in soils from a typical e-waste recycling region in southeast China. *Ecotoxicology, 24*(9), 1947-1960. doi:https://doi.org/10.1007/s10646-015-1532-7

Zheng, X., Xu, X., Yekeen, T. A., Zhang, Y., Chen, A., Kim, S. S., Kim N. Dietrich, K. N. D., Ho, S.-M., Lee, S.-A., Reponen, T., & Huo, X. (2016). Ambient Air Heavy Metals in PM2.5 and Potential Human Health Risk Assessment in an Informal Electronic-Waste Recycling Site of China. *Aerosol and Air Quality*

Research, 16(2), 388-397. doi:10.4209/aaqr.2014.11.0292

Zhuang, X. (2019). Chemical Hazards Associated With Treatment of Waste Electrical and Electronic Equipment. In *Electronic Waste Management and Treatment Technology* (pp. 311-334): Elsevier.

Appendix A

Photos of the soil sampling points

Figure A-1 The sampling point at the groundwater-well area (DY-GW-2)

Figure A-2 The sampling point at the e-waste dismantling site (DY-EW-1)

Figure A-3 The sampling point at open-burning site (DY-OB-1)

Figure A-4 The sampling point at the non-e-waste dismantling house (DY-NE-1)

Figure A-5 The sampling point at the paddy fields (PF)

Appendix B

Quality assurance and quality control

Table B-1 The results of parameters assessed in quality assurance and quality control

Heavy metal	**Recovery (%)**		**RSD (%)**	**LOD (mg/kg)**	**LOQ (mg/kg)**	**Regression equation**	**Coefficient (R^2)**
	Low	High					
As	79.37	88.06	0.97	0.172	0.575	Y=0.274x	1.000
Cd	108	117.39	1.43	0.013	0.042	Y=5.926x	0.999
Cr	94.89	102.41	4.67	3.892	12.97	Y=0.011x	0.999
Cu	82.59	90.77	5.07	1.058	3.529	Y=0.081x	0.998
Mn	83.68	90.12	6.75	1.594	5.315	Y=0.052x	0.998
Ni	85.68	94.11	1.10	0.025	0.085	Y=3.380x	1.000
Pb	85.36	91.22	2.55	0.102	0.340	Y=0.716x	0.994
Zn	88.24	90.76	1.87	0.007	0.026	Y=7.295x	0.999

Appendix C

Heavy metal concentrations in soils in the study area

Table C-1 Heavy metal concentrations in soil in dry season

no.	code	sampling site	soil layer	As		Cd		Cr		Cu	
				mean	SD	mean	SD	mean	SD	mean	SD
1	Control	control site	surface	11.036	0.224	0.645	0.009	29.118	2.728	9.376	1.000
			sub-surface	12.746	0.217	0.690	0.010	33.539	0.545	9.857	0.174
2	DY-EW-1	e-waste house	surface	6.012	0.704	0.490	0.072	24.093	1.908	22.007	1.142
			sub-surface	7.703	0.129	0.438	0.019	24.370	0.507	6.942	0.007
3	DY-NE-1	non-e-waste house	surface	9.852	0.498	0.346	0.025	14.100	1.258	0.785	0.129
			sub-surface	10.037	0.430	0.493	0.019	25.934	1.026	5.875	0.174
4	DY-OB-1	open-dumping site	surface	28.817	0.575	2.289	0.028	65.837	1.577	990.326	196.388
			sub-surface	27.586	2.436	1.655	0.134	56.456	3.457	140.296	124.356
5	DY-PF-1	paddy field	surface	21.493	0.656	1.113	0.051	40.745	1.823	25.942	0.774
			sub-surface	26.212	2.078	1.519	0.146	44.054	3.558	21.025	1.574
6	DY-PF-2	paddy field	surface	7.953	0.459	0.530	0.060	18.314	2.280	5.723	1.114
			sub-surface	18.820	0.775	0.777	0.031	28.996	0.942	7.760	0.539
7	DY-PF-3	paddy field	surface	29.501	1.685	1.699	0.108	49.490	1.927	91.130	1.765
			sub-surface	27.948	11.081	1.390	0.448	35.255	12.061	25.694	4.707
8	DY-PF-4	paddy field	surface	17.563	0.896	1.263	0.398	30.503	12.411	24.000	5.041
			sub-surface	20.363	0.704	1.109	0.063	25.855	2.950	12.101	1.026
9	DY-GW-1	Groundwater well	surface	9.501	0.118	0.532	0.014	11.474	1.078	<0.270	0.263
			sub-surface	6.415	0.209	0.424	0.016	9.097	0.901	<0.270	0.345
10	DY-GW-2	Groundwater well	surface	5.288	0.248	0.382	0.017	5.250	1.461	8.800	1.235
			sub-surface	6.439	0.408	0.455	0.036	7.652	0.979	<0.270	0.795
12	BP-EW-1	E-waste house	surface	5.236	0.355	0.286	0.033	14.594	1.415	60.987	38.603
			sub-surface	3.169	0.160	0.175	0.022	11.105	0.427	7.725	0.254
13	BP-NE-1	non-e-waste house	surface	8.329	2.071	0.345	0.111	19.222	2.374	36.852	54.497
			sub-surface	11.429	1.150	0.480	0.086	20.683	1.240	4.691	1.042
14	BP-OD-1	open-burning site	surface	11.177	0.169	0.507	0.026	21.043	0.051	245.412	198.252
			sub-surface	11.599	0.098	0.766	0.041	21.783	0.995	48.658	2.320
15	BP-PF-1	paddy field	surface	8.046	0.857	0.360	0.049	14.090	1.545	2.911	0.486
			sub-surface	14.618	2.433	0.607	0.137	20.124	1.100	3.197	0.474
16	BP-PF-2	paddy field	surface	5.839	0.059	0.230	0.004	12.141	0.965	2.348	0.078
			sub-surface	7.937	0.092	0.287	0.003	13.831	0.995	1.094	0.065
17	BP-GW-1	Groundwater well	surface	7.598	0.108	0.291	0.004	14.627	0.712	2.644	0.154
			sub-surface	8.475	0.268	0.331	0.019	13.276	0.453	2.602	1.384
18	BP-GW-2	Groundwater well	surface	8.706	0.354	0.342	0.015	14.835	1.025	5.377	0.357
			sub-surface	5.009	0.128	0.210	0.005	11.589	0.406	1.169	0.077

Table C-1 Heavy metal concentrations in soil in dry season (continued)

no	code	sampling site	soil layer	Mn		Ni		Pb		Zn	
				mean	SD	mean	SD	mean	SD	mean	SD
1	Control	control site	surface	66.027	4.218	6.527	0.131	13.219	0.065	5.683	0.165
			sub-surface	41.097	4.511	8.070	0.153	14.335	0.223	6.026	0.192
2	DY-EW-1	e-waste house	surface	97.456	7.539	5.642	0.182	19.245	1.666	71.883	32.455
			sub-surface	39.944	1.294	4.270	0.088	9.074	0.124	23.846	27.057
3	DY-NE-1	non-e-waste house	surface	36.299	2.583	3.559	0.255	10.042	0.555	3.335	0.524
			sub-surface	70.799	2.685	4.334	0.138	10.457	0.273	4.128	0.557
4	DY-OB-1	open-dumping site	surface	365.826	91.487	17.893	2.954	211.124	20.974	206.412	57.236
			sub-surface	210.691	24.904	13.802	0.838	58.109	16.945	48.812	2.274
5	DY-PF-1	paddy field	surface	200.270	4.735	12.328	0.320	25.706	0.508	15.570	0.202
			sub-surface	296.390	88.010	20.361	11.150	27.695	1.795	13.279	0.883
6	DY-PF-2	paddy field	surface	92.281	7.968	4.615	0.339	9.486	0.827	3.639	0.282
			sub-surface	55.850	4.011	8.587	0.715	18.202	0.643	8.763	0.354
7	DY-PF-3	paddy field	surface	296.288	28.534	14.822	1.203	44.034	1.102	27.225	0.551
			sub-surface	257.919	166.840	14.267	5.653	31.498	8.347	16.318	1.303
8	DY-PF-4	paddy field	surface	172.781	167.609	11.994	5.267	28.647	8.102	16.671	1.239
			sub-surface	140.205	10.996	10.198	0.237	23.577	0.676	12.094	0.093
9	DY-GW-1	Groundwater well	surface	85.868	4.630	5.363	0.289	10.175	0.210	7.201	0.429
			sub-surface	32.513	25.492	4.224	0.349	7.155	0.113	3.964	0.381
10	DY-GW-2	Groundwater well	surface	30.619	2.190	4.724	0.455	12.185	0.648	116.996	8.015
			sub-surface	49.480	3.319	4.253	0.174	8.872	0.549	36.145	1.288
12	BP-EW-1	E-waste house	surface	107.788	32.736	5.423	0.987	15.573	2.934	58.865	4.572
			sub-surface	101.359	4.230	2.861	0.465	8.423	1.995	14.301	0.211
13	BP-NE-1	non-e-waste house	surface	83.260	34.578	4.066	0.753	8.456	0.398	20.073	15.356
			sub-surface	121.886	16.696	5.980	0.790	12.794	0.913	6.763	0.516
14	BP-OD-1	open-burning site	surface	47.552	0.315	5.165	0.226	49.538	1.372	38.846	1.237
			sub-surface	86.499	1.443	5.608	0.081	28.557	2.599	62.056	3.171
15	BP-PF-1	paddy field	surface	63.961	7.324	3.903	0.260	8.170	0.510	4.503	0.431
			sub-surface	139.537	10.711	5.741	0.602	13.060	1.350	5.682	0.526
16	BP-PF-2	paddy field	surface	20.160	0.557	2.886	0.099	6.475	0.087	3.089	0.230
			sub-surface	47.715	0.487	4.008	0.080	7.654	0.102	2.882	0.055
17	BP-GW-1	Groundwater well	surface	139.022	1.103	4.478	0.033	8.757	0.137	7.914	0.306
			sub-surface	176.228	29.614	5.190	0.173	10.747	0.222	22.414	1.017
18	BP-GW-2	Groundwater well	surface	103.581	2.440	4.994	0.362	11.162	0.458	18.369	1.416
			sub-surface	70.959	2.097	2.906	0.133	5.952	0.050	5.924	0.346

Table C-2 Heavy metal concentrations in soil in wet season

no.	code	sampling site	soil layer	**As**		**Cd**		**Cr**		**Cu**	
				mean	**SD**	**mean**	**SD**	**mean**	**SD**	**mean**	**SD**
1	Control	control site	surface	11.405	0.423	0.621	0.031	18.489	1.973	6.416	0.398
			sub-surface	15.546	0.300	0.759	0.024	20.929	2.520	6.278	0.412
2	Ref	Reference	surface	9.733	0.133	0.448	0.012	11.610	0.772	6.016	0.324
			sub-surface	7.564	0.283	0.352	0.030	9.992	1.700	3.976	0.966
3	DY-NE-1	non-e-waste house	surface	10.217	0.347	0.455	0.025	3.750	1.559	<0.678	1.093
			sub-surface	9.843	0.492	0.457	0.017	3.622	1.530	<0.678	0.436
4	DY-GW-1	ground water well	surface	9.573	0.437	0.469	0.033	3.795	1.582	<0.678	0.432
			sub-surface	5.086	0.560	0.336	0.058	2.224	0.883	<0.678	0.774
5	DY-GW-2	ground water well	surface	4.482	0.424	0.357	0.079	1.030	0.704	11.966	2.005
			sub-surface	5.707	0.221	0.384	0.024	1.391	1.406	5.112	0.466
6	DY-EW-1	e-waste house#1	surface	9.111	0.598	0.651	0.154	3.675	0.032	50.249	36.586
			sub-surface	9.566	1.160	0.491	0.074	3.566	1.833	<0.678	0.390
7	DY-PF-1	paddy field	surface	11.839	2.547	0.739	0.177	5.680	0.751	<0.678	0.211
			sub-surface	31.114	1.011	1.726	0.005	15.865	0.966	6.425	0.057
8	DY-PF-2	paddy field	surface	7.873	0.304	0.462	0.040	0.366	0.730	8.495	0.695
			sub-surface	6.277	0.371	0.344	0.024	0.649	0.391	<0.678	0.529
9	DY-PF-3	paddy field	surface	13.089	0.639	0.706	0.055	25.209	1.830	50.931	3.259
			sub-surface	17.053	0.641	0.834	0.008	28.448	0.543	57.423	9.478
10	DY-PF-4	paddy field	surface	16.161	1.280	0.864	0.083	26.897	2.163	13.978	0.716
			sub-surface	39.076	2.838	1.716	0.116	40.642	2.092	21.447	1.963
11	DY-EW-2	e-waste house#2	surface	10.226	0.394	0.399	0.023	12.770	1.126	3.343	2.802
			sub-surface	6.771	0.301	0.248	0.018	10.342	0.383	<0.678	1.348
12	DY-OD-1	open-burning	surface	15.073	1.111	1.367	0.042	22.357	2.308	1725.450	1170.511
			sub-surface	25.798	1.325	10.930	0.698	45.305	10.683	24431.560	4926.293
13	BP-GW-1	Groundwater well	surface	5.320	0.185	0.124	0.011	3.927	0.538	<0.678	0.106
			sub-surface	4.749	0.180	<0.031	0.461	3.347	0.118	<0.678	0.636
14	BP-GW-2	Groundwater well	surface	12.620	0.520	0.460	0.019	20.713	1.291	3.094	0.579
			sub-surface	5.466	0.207	0.257	0.003	12.218	0.459	<0.678	2.568
15	BP-NE-1	non-e-waste house	surface	4.790	0.196	0.259	0.013	10.684	1.465	2.821	0.309
			sub-surface	4.988	0.652	0.250	0.034	10.505	1.462	<0.678	0.426
16	BP-PF-1	paddy field	surface	2.062	0.072	0.111	0.003	6.660	0.823	<0.678	0.354
			sub-surface	1.443	0.029	0.093	0.006	6.737	0.162	<0.678	0.076
17	BP-PF-2	paddy field	surface	2.010	0.099	0.112	0.012	6.243	0.859	7.938	1.048
			sub-surface	5.493	0.156	0.237	0.010	10.919	0.030	<0.678	0.145
18	BP-EW-1	E-waste house	surface	3.227	0.167	0.156	0.014	7.659	0.752	4.570	0.814
			sub-surface	1.622	1.765	0.113	0.009	6.773	0.319	<0.678	0.149
19	BP-OD-1	open-burning site	surface	10.584	1.053	0.786	0.118	15.352	2.233	54.173	1.606
			sub-surface	10.676	0.052	0.505	0.009	16.021	0.413	10.301	0.078

Table C-2 Heavy metal concentrations in soil in wet season (continued)

no.	code	sampling site	soil layer	Mn		Ni		Pb		Zn	
				Mean	SD	Mean	SD	Mean	SD	Mean	SD
1	Control	control site	surface	41.521	6.580	5.845	0.341	12.115	0.484	6.370	0.236
			sub-surface	30.772	7.501	8.103	0.328	15.213	0.173	7.172	0.141
2	Ref	Reference	surface	304.137	24.910	7.089	0.175	10.272	0.357	6.884	0.158
			sub-surface	108.350	10.059	5.500	0.223	7.679	0.322	5.636	0.437
3	DY-NE-1	non-e-waste house	surface	19.073	0.943	4.203	0.083	10.100	0.221	6.404	0.600
			sub-surface	19.164	2.227	3.989	0.122	9.603	0.344	5.799	0.359
4	DY-GW-1	ground water well	surface	56.575	3.484	4.646	0.212	10.052	0.281	7.779	0.086
			sub-surface	7.567	1.179	2.638	0.152	5.706	0.402	3.673	0.110
5	DY-GW-2	ground water well	surface	22.924	5.133	4.082	0.331	26.550	6.218	160.051	8.772
			sub-surface	33.600	1.370	3.809	0.338	21.132	2.237	56.666	0.796
6	DY-EW-1	e-waste house#1	surface	46.759	1.023	8.123	0.533	28.131	3.569	123.404	0.411
			sub-surface	14.379	1.242	3.848	0.315	9.886	0.539	8.358	0.402
7	DY-PF-1	paddy field	surface	24.223	5.881	9.352	6.041	12.778	1.605	7.247	0.334
			sub-surface	243.816	56.534	12.055	5.983	36.763	1.505	17.671	0.492
8	DY-PF-2	paddy field	surface	16.328	1.548	4.772	0.334	16.778	2.496	15.317	1.381
			sub-surface	18.620	2.282	3.904	0.203	8.670	0.505	5.771	0.646
9	DY-PF-3	paddy field	surface	239.438	27.965	8.513	1.225	24.270	0.773	16.276	0.593
			sub-surface	275.036	22.072	9.891	0.497	26.368	0.296	16.475	0.896
10	DY-PF-4	paddy field	surface	112.443	9.823	8.891	0.478	22.673	1.788	13.620	0.942
			sub-surface	548.054	67.840	23.186	1.223	50.318	5.498	24.082	0.704
11	DY-EW-2	e-waste house#2	surface	168.102	28.327	6.267	0.162	12.761	1.203	19.035	0.778
			sub-surface	125.966	11.869	4.337	0.317	7.848	0.290	6.021	0.358
12	DY-OD-1	open-burning	surface	183.811	45.558	10.600	0.442	108.583	4.883	170.215	12.754
			sub-surface	338.127	31.340	94.173	11.007	2700.774	45.164	1735.856	146.269
13	BP-GW-1	Groundwater well	surface	168.289	26.861	4.774	0.383	10.358	0.550	11.528	0.290
			sub-surface	338.519	76.062	4.339	0.016	9.140	1.122	15.801	0.135
14	BP-GW-2	Groundwater well	surface	175.742	19.645	6.808	0.218	15.728	0.703	19.551	0.984
			sub-surface	72.211	58.500	3.540	0.173	8.691	0.165	12.625	0.442
15	BP-NE-1	non-e-waste house	surface	121.176	5.029	3.102	0.172	7.909	0.322	26.942	1.453
			sub-surface	83.147	5.301	3.281	0.425	7.286	0.625	8.397	0.518
16	BP-PF-1	paddy field	surface	0.823	2.100	1.633	0.048	4.346	0.060	1.835	0.901
			sub-surface	0.162	0.266	1.129	0.124	2.309	0.060	0.648	0.197
17	BP-PF-2	paddy field	surface	14.093	2.911	1.215	0.009	5.089	0.102	2.869	0.177
			sub-surface	37.682	3.035	2.771	0.209	6.216	0.051	5.083	0.290
18	BP-EW-1	E-waste house	surface	99.522	15.904	2.326	0.275	6.811	0.417	19.469	2.060
			sub-surface	62.092	1.281	1.774	0.050	4.733	0.081	5.390	1.902
19	BP-OD-1	open-burning site	surface	135.820	6.649	6.448	0.790	27.528	2.515	88.892	9.549
			sub-surface	104.178	15.062	5.023	0.082	14.788	0.058	34.624	7.068

Appendix D

Total heavy metal contributions in the soil and enrichment factor calculation results

Table D-1 Heavy metal contribution in surface soils

HM contribution (%)								
Sites	**As**	**Cd**	**Cr**	**Cu**	**Mn**	**Ni**	**Pb**	**Zn**
reference	2.73	0.13	3.26	1.69	85.39	1.99	2.88	1.93
control	9.18	0.52	19.48	6.46	44.00	5.06	10.37	4.93
non-e-waste	6.76	0.29	9.72	8.31	52.90	3.04	7.43	11.56
e-waste	2.91	0.17	5.40	12.14	44.71	2.39	7.10	25.18
groundwater well	4.35	0.20	5.21	2.25	53.93	2.75	7.23	24.08
paddy fields	6.25	0.36	10.29	10.19	54.57	3.70	9.08	5.57
open-burning	1.34	0.10	2.55	61.73	15.01	0.82	8.12	10.33

Table D-2 Heavy metal contribution in subsurface soils

HM contribution (%)								
site	**As**	**Cd**	**Cr**	**Cu**	**Mn**	**Ni**	**Pb**	**Zn**
reference	5.08	0.24	6.70	2.67	72.69	3.69	5.15	3.78
control	12.24	0.63	23.57	6.98	31.09	7.00	12.78	5.71
non-e-waste	7.44	0.34	12.45	2.31	60.48	3.60	8.23	5.14
e-waste	5.14	0.26	10.01	2.80	61.29	3.05	7.13	10.33
groundwater well	4.06	0.21	5.21	0.87	66.91	2.65	6.63	13.47
paddy fields	6.73	0.33	8.44	4.89	64.13	3.61	7.85	4.01
open-burning	0.25	0.05	0.46	81.02	2.43	0.39	9.22	6.19

Table D-3 Enrichment factor of heavy metal in surface soils

Seasons	**code**	**As**	**Cd**	**Cr**	**Cu**	**Ni**	**Pb**	**Zn**
Dry	control	5.22	6.62	11.55	7.18	4.24	5.93	3.80
Wet	control	8.58	10.15	11.66	7.81	6.04	8.64	6.78
	ref	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Dry	DY-NE-1	8.48	6.46	10.18	1.09	4.21	8.19	4.06
	BP-NE-1	3.13	2.81	6.05	22.37	2.09	3.01	10.65
Wet	DY-NE-1	16.74	16.17	5.15	0.90	9.45	15.68	14.83
	DY-NE-1	1.24	1.45	2.31	1.18	1.10	1.93	9.82
Dry	DY-EW-1	1.93	3.41	6.48	11.42	2.48	5.85	32.59
	BP-EW-1	1.52	1.80	3.55	28.60	2.16	4.28	24.13
Wet	DY-EW-1	6.09	9.44	2.06	54.32	7.45	17.81	116.60
	DY-EW-2	1.90	1.61	1.99	1.01	1.60	2.25	5.00
	BP-EW-1	1.01	1.06	2.02	2.32	1.00	2.03	8.64
Dry	DY-GW-1	3.46	4.20	3.50	0.08	2.68	3.51	3.71
	DY-GW-2	5.40	8.45	4.49	14.53	6.62	11.78	168.82
	BP-GW-1	1.71	1.42	2.76	0.96	1.38	1.87	2.52
	BP-GW-2	2.63	2.24	3.75	2.62	2.07	3.19	7.84

Seasons	code	As	Cd	Cr	Cu	Ni	Pb	Zn
Wet	DY-GW-1	5.29	5.63	1.76	0.30	3.52	5.26	6.07
	DY-GW-2	6.11	10.55	1.18	26.39	7.64	34.29	308.47
	BP-GW-1	0.99	0.50	0.61	0.10	1.22	1.82	3.03
	BP-GW-2	2.24	1.77	3.09	0.89	1.66	2.65	4.92
Dry	DY-PF-1	3.35	3.77	5.33	6.55	2.64	3.80	3.43
	DY-PF-2	2.69	3.89	5.20	3.13	2.15	3.04	1.74
	DY-PF-3	3.11	3.89	4.38	15.55	2.15	4.40	4.06
	DY-PF-4	3.18	4.96	4.62	7.02	2.98	4.91	4.26
	BP-PF-1	3.93	3.82	5.77	2.30	2.62	3.78	3.11
	BP-PF-2	9.05	7.74	15.78	5.89	6.14	9.51	6.77
Wet	DY-PF-1	15.27	20.69	6.14	0.71	16.56	15.62	13.22
	DY-PF-2	15.07	19.20	0.59	26.30	12.54	30.42	41.44
	DY-PF-3	1.71	2.00	2.76	10.75	1.53	3.00	3.00
	DY-PF-4	4.49	5.21	6.27	6.28	3.39	5.97	5.35
	BP-PF-1	78.34	91.56	212.10	20.83	85.15	156.43	98.59
	BP-PF-2	4.46	5.38	11.60	28.47	3.70	10.69	8.99
Dry	DY-OD-1	2.46	4.24	4.71	136.85	2.10	17.09	24.93
	BP-OD-1	7.35	7.24	11.59	260.89	4.66	30.84	36.09
	DY-OD-1	2.56	5.05	3.19	474.53	2.47	17.49	40.91
	BP-OD-1	2.44	3.93	2.96	20.16	2.04	6.00	28.92

Table D-4 Enrichment factor of heavy metal in subsurface soils

Seasons	code	As	Cd	Cr	Cu	Ni	Pb	Zn
Dry	control	4.44	5.17	8.85	6.54	3.87	4.92	2.82
Wet	control	7.24	7.60	7.37	5.56	5.19	6.98	4.48
	ref	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Dry	DY-NE-1	2.03	2.14	3.97	2.26	1.21	2.08	1.12
	BP-NE-1	1.34	1.21	1.84	1.05	0.97	1.48	1.07
Wet	DY-NE-1	7.36	7.35	2.05	0.48	4.10	7.07	5.82
	DY-NE-1	0.86	0.93	1.37	0.11	0.78	1.24	1.94
Dry	DY-EW-1	2.76	3.38	6.62	4.74	2.11	3.21	11.48
	BP-EW-1	0.45	0.53	1.19	2.08	0.56	1.17	2.71
Wet	DY-EW-1	9.53	10.52	2.69	0.64	5.27	9.70	11.17
	DY-EW-2	0.77	0.61	0.89	0.07	0.68	0.88	0.92
	BP-EW-1	0.37	0.56	1.18	0.15	0.56	1.08	1.67
Dry	DY-GW-1	2.83	4.02	3.03	0.11	2.56	3.11	2.34
	DY-GW-2	1.86	2.83	1.68	0.07	1.69	2.53	14.04
	BP-GW-1	0.69	0.58	0.82	0.40	0.58	0.86	2.45
	BP-GW-2	1.01	0.91	1.77	0.45	0.81	1.18	1.61
Wet	DY-GW-1	9.63	13.68	3.19	1.22	6.87	10.64	9.33
	DY-GW-2	2.43	3.52	0.45	4.15	2.23	8.87	32.42

Seasons	code	As	Cd	Cr	Cu	Ni	Pb	Zn
	BP-GW-1	0.20	0.01	0.11	0.03	0.25	0.38	0.90
	BP-GW-2	1.08	1.09	1.83	0.13	0.97	1.70	3.36
Dry	DY-PF-1	1.27	1.58	1.61	1.93	1.35	1.32	0.86
	DY-PF-2	4.83	4.29	5.63	3.79	3.03	4.60	3.02
	DY-PF-3	1.55	1.66	1.48	2.71	1.09	1.72	1.22
	DY-PF-4	2.08	2.44	2.00	2.35	1.43	2.37	1.66
	BP-PF-1	1.50	1.34	1.56	0.62	0.81	1.32	0.78
	BP-PF-2	2.38	1.85	3.14	0.62	1.65	2.26	1.16
Wet	DY-PF-1	1.83	2.18	0.71	0.72	0.97	2.13	1.39
	DY-PF-2	4.83	5.69	0.38	0.50	4.13	6.57	5.96
	DY-PF-3	0.89	0.93	1.12	5.69	0.71	1.35	1.15
	DY-PF-4	1.02	0.96	0.80	1.07	0.83	1.30	0.84
	BP-PF-1	127.39	176.14	450.07	56.92	137.05	200.78	76.75
	BP-PF-2	2.09	1.93	3.14	0.25	1.45	2.33	2.59
Dry	DY-OD-1	1.88	2.42	2.91	18.15	1.29	3.89	4.45
	BP-OD-1	1.92	2.73	2.73	15.33	1.28	4.66	13.79
	DY-OD-1	1.09	9.96	1.45	1968.99	5.49	112.71	98.70
	BP-OD-1	1.47	1.49	1.67	2.69	0.95	2.00	6.39

Appendix E

Heavy metal fractionation in the study area

Table E-1 Percentage of As fraction in the study area

As		Surface			Subsurface		
Sites	**season**	**F1**	**F2**	**F3**	**F1**	**F2**	**F3**
Control	Dry	7.94	11.96	80.09	7.05	11.08	81.87
	Wet	7.58	8.74	83.68	6.75	7.61	85.64
GW	Dry	7.70	6.63	85.67	9.23	7.42	83.36
	Wet	7.85	5.39	86.77	9.35	7.25	83.41
EW	Dry	6.99	5.31	87.70	6.75	5.76	87.48
	Wet	9.00	7.34	83.67	6.38	4.47	89.15
PF	Dry	9.59	13.08	77.33	8.44	15.51	76.05
	Wet	9.18	11.28	79.54	8.81	13.05	78.14
OB	Dry	16.78	22.73	60.48	10.16	15.12	74.72
	Wet	11.70	21.58	66.71	21.17	64.32	14.51

Table E-2 Percentage of Cd fraction in the study area

Cd		Surface			Subsurface		
Sites	**season**	**F1**	**F2**	**F3**	**F1**	**F2**	**F3**
Control	Dry	4.13	75.89	19.98	5.49	67.93	26.58
	Wet	6.08	64.45	29.47	9.44	44.84	45.72
GW	Dry	6.13	64.18	29.69	6.27	63.34	30.38
	Wet	6.78	60.38	32.84	5.34	68.82	25.84
EW	Dry	40.52	30.01	29.47	10.49	38.68	50.82
	Wet	30.21	43.24	26.55	12.26	28.36	59.38
PF	Dry	20.25	64.17	15.58	4.13	75.88	19.99
	Wet	4.98	70.88	24.14	4.56	73.37	22.07
OB	Dry	52.01	45.18	2.81	24.74	64.14	11.12
	Wet	50.74	46.19	3.07	42.43	49.78	7.78

Table E-3 Percentage of Cr fraction in the study area

Cr		Surface			Subsurface		
Sites	**season**	**F1**	**F2**	**F3**	**F1**	**F2**	**F3**
Control	Dry	7.53	85.84	6.63	3.40	87.90	8.70
	Wet	34.10	52.35	13.56	19.19	63.30	17.51
GW	Dry	11.76	50.60	37.64	18.33	47.79	33.88
	Wet	11.65	48.54	39.81	14.81	43.52	41.67
EW	Dry	23.38	40.49	36.13	19.06	38.95	41.99
	Wet	22.62	22.07	55.31	24.47	26.13	49.40
PF	Dry	21.71	29.93	48.36	17.68	40.99	41.34
	Wet	16.21	51.07	32.72	12.45	34.44	53.11
OB	Dry	7.24	30.36	62.40	3.05	6.20	90.75
	Wet	6.49	30.47	63.04	10.16	49.54	40.30

Table E-4 Percentage of Cu fraction in the study area

Cu		**Surface**			**Subsurface**		
Sites	season	F1	F2	F3	**F1**	**F2**	**F3**
Control	Dry	8.191	85.605	6.204	9.621	83.091	7.288
	Wet	9.221	83.795	6.984	11.802	79.259	8.939
GW	Dry	31.084	54.569	14.348	17.347	62.245	20.408
	Wet	47.619	34.586	17.794	28.767	47.358	23.875
EW	Dry	43.749	42.075	14.176	34.215	56.753	9.032
	Wet	63.793	27.567	8.640	36.364	48.485	15.152
PF	Dry	28.551	54.126	17.323	7.459	78.947	13.594
	Wet	32.000	58.320	9.680	19.280	69.266	11.454
OB	Dry	41.410	23.217	35.373	30.983	26.711	42.306
	Wet	32.693	25.150	42.157	26.898	57.607	15.495

Table E-5 Percentage of Mn fraction in the study area

Mn		**Surface**			**Subsurface**		
Sites	season	F1	F2	F3	F1	F2	F3
Control	Dry	39.22	56.29	4.49	55.35	40.30	4.35
	Wet	39.27	52.90	7.83	29.85	66.64	3.51
GW	Dry	74.65	22.82	2.53	49.72	46.44	3.85
	Wet	72.08	25.13	2.79	48.61	48.04	3.35
EW	Dry	56.59	40.00	3.41	77.92	17.19	4.89
	Wet	70.17	25.25	4.58	64.70	23.48	11.81
PF	Dry	28.72	67.39	3.89	8.82	86.87	4.31
	Wet	28.25	68.24	3.50	15.70	79.09	5.21
OB	Dry	41.54	53.07	5.39	35.19	56.24	8.57
	Wet	52.77	40.86	6.38	31.29	62.47	6.24

Table E-6 Percentage of Ni fraction in the study area

Ni		**Surface**			**Subsurface**		
Sites	season	F1	F2	F3	F1	F2	F3
Control	Dry	16.99	35.55	47.46	21.39	27.18	51.44
	Wet	24.82	27.24	47.93	20.79	21.28	57.92
GW	Dry	45.93	25.92	28.15	27.56	37.66	34.78
	Wet	44.70	27.46	27.84	29.34	37.12	33.55
EW	Dry	47.26	20.42	32.32	16.78	4.93	78.29
	Wet	40.09	32.52	27.39	22.96	13.72	63.32
PF	Dry	13.15	30.50	56.36	11.30	35.90	52.80
	Wet	13.50	32.09	54.41	11.57	31.01	57.42
OB	Dry	35.65	31.81	32.54	16.89	31.32	51.79
	Wet	33.19	32.63	34.18	23.34	63.84	12.82

Table E-7 Percentage of Pb fraction in the study area

Pb		**surface**			**subsurface**		
Sites	**season**	**F1**	**F2**	**F3**	**F1**	**F2**	**F3**
Control	Dry	3.878	71.019	25.103	4.13	71.57	24.29
	Wet	3.960	79.195	16.845	2.18	73.17	24.64
GW	Dry	15.853	82.797	1.350	6.83	90.47	2.71
	Wet	25.103	72.713	2.184	17.21	79.62	3.17
EW	Dry	15.235	75.993	8.772	2.55	77.45	20.00
	Wet	21.493	71.100	7.407	1.93	81.79	16.28
PF	Dry	4.327	77.733	17.939	2.67	75.55	21.78
	Wet	4.658	80.469	14.873	3.48	78.99	17.52
OB	Dry	23.532	60.607	15.861	6.03	76.75	17.22
	Wet	29.813	58.143	12.044	31.45	54.54	14.01

Table E-8 Percentage of Zn fraction in the study area

Zn		**Surface**			**Subsurface**		
Sites	season	F1	F2	F3	F1	F2	F3
Control	Dry	36.50	31.23	32.27	45.48	28.58	25.94
	Wet	48.40	20.69	30.90	47.31	20.75	31.94
GW	Dry	77.56	15.27	7.16	55.97	28.66	15.37
	Wet	77.11	17.08	5.81	60.07	25.81	14.12
EW	Dry	60.39	26.72	12.89	47.56	23.23	29.21
	Wet	64.74	25.33	9.94	43.61	27.16	29.23
PF	Dry	37.02	38.56	24.42	21.46	42.77	35.78
	Wet	33.42	40.94	25.64	35.63	38.29	26.08
OB	Dry	68.80	21.37	9.82	50.74	31.13	18.14
	Wet	72.91	19.38	7.71	45.21	47.51	7.28

Appendix F

Potential Ecological Risk Index (PERI) and Risk Assessment Code (RAC)

Table F-1 Ecological risk (Er) and PERI of heavy metals in surface soils in the study area

Seasons	code	As	Cd	Cr	Cu	Mn	Ni	Pb	Zn	PERI
Dry	control	11.34	43.15	5.02	7.79	0.22	4.60	6.43	0.83	79.37
	control	11.72	41.56	3.18	5.33	0.14	4.12	5.90	0.93	72.88
Dry	DY-NE-1	10.12	23.15	2.43	0.65	0.12	2.51	4.89	0.48	44.35
	BP-NE-1	8.56	23.09	3.31	30.63	0.27	2.87	4.12	2.92	75.76
Wet	DY-NE-1	10.50	30.43	0.65	0.28	0.06	2.96	4.92	0.93	50.73
	DY-NE-1	4.92	17.35	1.84	2.34	0.40	2.19	3.85	3.91	36.80
Dry	DY-EW-1	6.18	32.82	4.15	18.29	0.32	3.98	9.37	10.44	85.54
	BP-EW-1	5.38	19.13	2.51	50.68	0.35	3.82	7.58	8.55	98.01
Wet	DY-EW-1	9.36	43.53	0.63	41.76	0.15	5.73	13.69	17.93	132.78
	DY-EW-2	10.51	26.70	2.20	2.78	0.55	4.42	6.21	2.77	56.13
	BP-EW-1	3.32	10.41	1.32	3.80	0.33	1.64	3.32	2.83	26.96
Dry	DY-GW-1	9.76	35.57	1.98	0.11	0.28	3.78	4.95	1.05	57.48
	DY-GW-2	5.43	25.53	0.90	7.31	0.10	3.33	5.93	17.00	65.54
	BP-GW-1	7.81	19.48	2.52	2.20	0.46	3.16	4.26	1.15	41.03
	BP-GW-2	8.94	22.87	2.56	4.47	0.34	3.52	5.43	2.67	50.81
Wet	DY-GW-1	9.84	31.39	0.65	0.28	0.19	3.28	4.89	1.13	51.65
	DY-GW-2	4.60	23.86	0.18	9.94	0.08	2.88	12.92	23.25	77.72
	BP-GW-1	5.47	8.30	0.68	0.28	0.55	3.37	5.04	1.67	25.36
	BP-GW-2	12.97	30.75	3.57	2.57	0.58	4.80	7.66	2.84	65.73
Dry	DY-PF-1	22.08	74.49	7.02	21.56	0.66	8.69	12.51	2.26	149.27
	DY-PF-2	8.17	35.44	3.15	4.76	0.30	3.25	4.62	0.53	60.23
	DY-PF-3	30.31	113.65	8.53	75.73	0.97	10.45	21.43	3.95	265.03
	DY-PF-4	18.05	84.49	5.25	19.95	0.57	8.46	13.94	2.42	153.13
	BP-PF-1	8.27	24.08	2.43	2.42	0.21	2.75	3.98	0.65	44.78
	BP-PF-2	6.00	15.40	2.09	1.95	0.07	2.04	3.15	0.45	31.14
Wet	DY-PF-1	12.16	49.43	0.98	0.28	0.08	6.60	6.22	1.05	76.80
	DY-PF-2	8.09	30.93	0.06	7.06	0.05	3.37	8.17	2.23	59.95
	DY-PF-3	13.45	47.25	4.34	42.33	0.79	6.00	11.81	2.36	128.34
	DY-PF-4	16.60	57.83	4.63	11.62	0.37	6.27	11.04	1.98	110.34
	BP-PF-1	2.12	7.43	1.15	0.28	0.00	1.15	2.12	0.27	14.51
	BP-PF-2	2.07	7.47	1.08	6.60	0.05	0.86	2.48	0.42	21.01
Dry	DY-OD-1	29.61	153.13	11.34	823.02	1.20	12.62	102.76	29.98	1163.67
	BP-OD-1	11.48	33.94	3.62	203.95	0.16	3.64	24.11	5.64	286.55
	DY-OD-1	15.49	91.48	3.85	1433.95	0.60	7.48	52.85	24.73	1630.43
	BP-OD-1	10.87	52.61	2.64	45.02	0.45	4.55	13.40	12.91	142.45

Table F-2 Ecological risk (Er) and PERI of heavy metals in subsurface soils in the study area

Seasons	**code**	**As**	**Cd**	**Cr**	**Cu**	**Mn**	**Ni**	**Pb**	**Zn**	**PERI**
Dry	control	16.85	58.86	6.71	12.39	0.38	7.34	9.33	1.07	112.94
	control	20.55	64.74	4.19	7.89	0.28	7.37	9.91	1.27	116.21
Dry	DY-NE-1	13.27	42.03	5.19	7.39	0.65	3.94	6.81	0.73	80.02
	BP-NE-1	15.11	40.94	4.14	5.90	1.12	5.44	8.33	1.20	82.18
Wet	DY-NE-1	13.01	39.00	0.72	0.43	0.18	3.63	6.25	1.03	64.25
	DY-NE-1	6.59	21.33	2.10	0.43	0.77	2.98	4.74	1.49	40.44
Dry	DY-EW-1	10.18	37.34	4.88	8.73	0.37	3.88	5.91	4.23	75.52
	BP-EW-1	4.19	14.93	2.22	9.71	0.94	2.60	5.48	2.54	42.61
Wet	DY-EW-1	12.65	41.89	0.71	0.43	0.13	3.50	6.44	1.48	67.22
	DY-EW-2	8.95	21.13	2.07	0.43	1.16	3.94	5.11	1.07	43.86
	BP-EW-1	2.14	9.61	1.36	0.43	0.57	1.61	3.08	0.96	19.76
Dry	DY-GW-1	8.48	36.17	1.82	0.17	0.30	3.84	4.66	0.70	56.14
	DY-GW-2	8.51	38.82	1.53	0.17	0.46	3.87	5.78	6.41	65.55
	BP-GW-1	11.20	28.21	2.66	3.27	1.63	4.72	7.00	3.98	62.66
	BP-GW-2	6.62	17.93	2.32	1.47	0.65	2.64	3.88	1.05	36.56
Wet	DY-GW-1	6.72	28.67	0.45	0.43	0.07	2.40	3.72	0.65	43.10
	DY-GW-2	7.54	32.79	0.28	6.43	0.31	3.46	13.76	10.05	74.63
	BP-GW-1	6.28	1.32	0.67	0.43	3.12	3.94	5.95	2.80	24.52
	BP-GW-2	7.23	21.88	2.45	0.43	0.67	3.22	5.66	2.24	43.76
Dry	DY-PF-1	34.65	129.50	8.82	26.44	2.74	18.51	18.03	2.36	241.04
	DY-PF-2	24.88	66.28	5.80	9.76	0.52	7.81	11.85	1.55	128.45
	DY-PF-3	36.95	118.50	7.06	32.31	2.38	12.97	20.51	2.90	233.57
	DY-PF-4	26.92	94.53	5.18	15.22	1.29	9.27	15.35	2.15	169.91
	BP-PF-1	19.32	51.79	4.03	4.02	1.29	5.22	8.50	1.01	95.18
	BP-PF-2	10.49	24.49	2.77	1.38	0.44	3.64	4.98	0.51	48.71
Wet	DY-PF-1	41.13	147.22	3.18	8.08	2.25	10.96	23.94	3.14	239.89
	DY-PF-2	8.30	29.34	0.13	0.43	0.17	3.55	5.65	1.02	48.59
	DY-PF-3	22.54	71.15	5.69	72.21	2.54	8.99	17.17	2.92	203.23
	DY-PF-4	51.66	146.31	8.13	26.97	5.06	21.08	32.77	4.27	296.24
	BP-PF-1	1.91	7.92	1.35	0.43	0.00	1.03	1.50	0.11	14.25
	BP-PF-2	7.26	20.19	2.19	0.43	0.35	2.52	4.05	0.90	37.88
Dry	DY-OD-1	36.47	141.14	11.30	176.42	1.94	12.55	37.84	8.66	426.33
	BP-OD-1	15.33	65.33	4.36	61.19	0.80	5.10	18.60	11.01	181.72
	DY-OD-1	34.11	932.10	9.07	30723.00	3.12	85.61	1758.62	308.00	33853.63
	BP-OD-1	14.11	43.05	3.21	12.95	0.96	4.57	9.63	6.14	94.62

Table F-3 Percentage of F1 of heavy metals in the study area

Sites	Seasons	Code	layers	Percentage of F1 (%)							
				As	**Cd**	**Cr**	**Cu**	**Mn**	**Ni**	**Pb**	**Zn**
Control	control	Dry	surface	7.37	0.99	6.14	5.65	27.36	2.87	1.27	36.30
			subsurface	5.56	0.93	1.83	5.37	27.33	2.91	0.83	39.22
		Wet	surface	6.23	1.03	21.42	8.25	30.49	4.38	0.85	46.20
			subsurface	4.02	0.84	8.24	8.43	24.40	3.00	0.34	37.25
EW	DY-EW-1	Dry	surface	10.94	8.69	14.58	31.25	42.46	21.25	9.75	54.41
			subsurface	7.40	1.46	16.38	21.88	56.74	12.89	0.56	8.79
		Wet	surface	8.78	5.42	22.85	59.86	76.03	26.70	18.28	61.96
			subsurface	5.60	1.30	91.88	35.40	64.02	6.53	0.39	41.29
PF	DY-PF-3	Dry	surface	3.24	2.23	9.62	23.72	23.48	3.60	2.29	19.82
			subsurface	3.45	0.45	13.55	5.06	6.81	3.80	0.96	6.59
		Wet	surface	6.83	0.91	11.08	15.09	17.54	4.59	2.12	23.09
			subsurface	5.28	0.77	8.81	8.63	9.98	4.03	1.40	21.33
GW	DY-GW-2	Dry	surface	11.82	1.68	63.74	98.52	74.60	19.42	9.64	82.98
			subsurface	12.68	1.41	72.62	17.26	48.70	11.39	2.25	55.97
		Wet	surface	15.56	1.79	11.65	47.63	72.07	23.50	7.41	76.04
			subsurface	15.43	1.66	14.78	28.76	73.39	15.61	14.39	59.73
OD	DY-OD-1	Dry	surface	6.82	25.09	22.45	34.55	20.99	17.74	21.38	58.94
			subsurface	3.56	4.17	7.44	23.13	18.37	5.46	2.27	40.12
		Wet	surface	7.82	37.52	58.94	23.85	24.96	18.43	25.77	68.66
			subsurface	36.99	63.38	10.16	13.38	16.85	19.30	22.78	25.76

VITA

NAME	Nisakorn Amphalop
DATE OF BIRTH	03 Aug 1996
PLACE OF BIRTH	Saraburi, Thailand
INSTITUTIONS ATTENDED	Bachelor of Science in Environmental Science, Chulalongkorn University Master of Science in Hazardous Substance and Environmental Management, Chulalongkorn University
HOME ADDRESS	74 Moo.10 Tandaew subdistrict, Kaeng-Khoi district, Saraburi Province, 18110
PUBLICATION	Amphalop, N., Prueksasit, T., & Assawadithalerd, M. (2020). Spatial Variation of Heavy Metals Contamination in Soil at E-waste Dismantling Site, Buriram Province, Thailand. Paper presented at the International Conference on Sustainable Development of Water and Environment. Amphalop, N., Suwantarat, N., Prueksasit, T., Yachusri, C., & Srithongouthai, S. (2020). Ecological risk assessment of arsenic, cadmium, copper, and lead contamination in soil in e-waste separating household area, Buriram province, Thailand. Environmental Science and Pollution Research, 1-16.
AWARD RECEIVED	Best Paper Award on 2020 The 3rd International Conference on Sustainable Development of Water and Environment (ICSDWE 2020)