A Perception-Response-Evaluation (PRAVE) framework for societal-physical-based risk decision-making: A case of endocrine disrupting surfactants contamination in Vietnam

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ABSTRACT

A Perception-Response-Evaluation (PRAVE) framework for societal-physical-based risk decision-making: A case of endocrine disrupting surfactants contamination in Vietnam

Endocrine disrupting contamination has been recognized to be a serious problem in modern societies over the last two decades. Nonylphenol and nonylphenol ethoxylates, known to have endocrine disrupting effects, have been using as surfactants in numerous industries, such as cleansing, textile, personal care products, pesticides, paper, and plastic. The chemicals are released into the environment mainly in form of domestic sewage and industrial effluent discharge. Recent studies have reported alarming levels of nonylphenol in the urban watercourses across Vietnam, which raises concerns about the ecological and health risk linked to endocrine disrupting surfactants (EDSs). Whilst a risk management framework for endocrine disrupting chemicals is lacking in Vietnam, and current risk management schemes merely rely on technical (or physical) risk assessments, this dissertation postulates an EDSs risk decision-making framework based on the integration of societal and physical perspectives (PRAVE framework). The research goal was operationalized in four studies that employed both types of qualitative and quantitative approaches.

Study 1 examined societal perspectives and self-protective behavior towards EDSs. Data was collected via a questionnaire survey and analyzed by structural equation modelling. Study 2 proposed an integrated modelling framework for evaluating non-carcinogenic health risks from nonylphenol contaminated food consumption. This was achieved by using ecological models which outputs were subjected to an exposure and risk evaluation. Study 3 proposed a firm behavior model based on a qualitative analysis of the Vietnamese textile firm's response to the need of nonylphenol and nonylphenol ethoxylates restriction. Study 4 assessed the distribution and removal of nonylphenol ethoxylates and nonylphenol from textile wastewater, where a cotton and a synthetic fiber factory were investigated and compared.

The first main findings were that: 1) the public judgments of EDSs risk were uncertainty-laden; 2) the patterns of risk perception and self-protection between pregnant women and young mothers and the remaining people were not significantly different; 3) perceived EDSs risk had no effect on the public habit of riverine fish consumption, whilst it was demonstrated that consuming EDS contaminated riverine foods posed a health risk to a large proportion of population; 4) the frequency of consuming EDS contaminated riverine foods was a key factor linked with health risk. This suggests immediate actions in order to tackle EDSs risk in urban cities. Accordingly, a social policy for raising awareness of EDCs risk among the public with a focus on pregnant women and young mothers is needed. To do so, building capacity for communicating and managing EDSs risk among governmental and non-governmental institutions, as well as conducting a comprehensive

EDSs risk assessment covering a wide range of food and drink is essential. Since the EDSs contamination in Ho Chi Minh city's canals has been found to have negative impact on the river in the adjacent province, the moral and legal responsibilities of the "polluter" towards the "sufferer" should be recognized in the legal system of environmental protection of Vietnam. Secondly, the study revealed that foreign regulations on EDSs restriction are able to impact positively the environment through changing the EDSs consumption and discharge behavior among the textile firms. Therefore, the roles of intermediary industrial/trade organizations in offering support to private textile firms in the form of useful information (e.g., regulatory and market changes, exporting requirements, technological and chemical innovations) are highlighted. Thirdly, there is a potency of continuing use and discharge of EDSs into the environment among the domestic textile firms, whereas only well-designed and controlled coagulation-activated sludge processes can yield promising EDSs removal for textile wastewater. Hence, assessing the available wastewater treatment facilities of potential industries with regards to EDSs removal is needed, particularly focusing on those discharging directly into the environment. Additionally, integrating NP/NPEOs into the current standards for sewage and industrial effluent discharge and for surface water is recommended, whilst long-term strategies should aim to restrict the use of EDSs in manufacturing industries and products.

EDCs risk management in Vietnam could be kick-started by applying PRAVE framework which allows to design comprehensive strategies by mediating scientific and societal perspectives. The decision-making process could be supported by sub-models of ecological-health risk evaluation, risk perception-self protective response, and firm's perception-behavior. Additionally, guidelines for dietary advice regarding riverine food product consumption and for assessment of wastewater treatment facilities regarding EDSs removal could be derived. The four studies in this dissertation, each contributes knowledge of one sphere of EDSs risk management, together have demonstrated the value and utility of PRAVE framework originally postulated by this dissertation.

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CHAPTER 1

INTRODUCTION

This dissertation proposes an integrated framework for managing the ecological and health risks of endocrine disrupting contamination. The framework comprises two domains of knowledge that complementarily support risk communication at local level and decision-making at administrational level: 1) Risk Perception and Response, and 2) Risk Evaluation. It is hereinafter referred to as PRAVE framework. On the basis of PRAVE, four case studies have been conducted with major contributions:

1) A quantitative study on the dynamic of endocrine disrupting surfactants (EDSs) risk perception and self-protective behavior among the lay and experienced public in the urban cities of Ho Chi Minh and Da Nang, Vietnam *(a societal reflection)*:

   The study sheds light to better understand the roles of three types of knowledge that distinguish by experience, and the mediating role of perceived uncertainty in explaining EDSs risk perception, acceptability, and self-protective response. The findings are essential for risk education and risk communication.

2) A modelling of EDS distribution and bioaccumulation for a river adjacent to the polluted city of Ho Chi Minh and health risk from consuming riverine foods *(a physical reflection)*:

   The study presents an integrated framework for screening ecological risk and human health risk of EDS contaminated food consumption that is useful in places where field-measured data is lacking. The modified method of evaluating non-carcinogenic health risk from EDS contaminated food consumption advances the current method by coupling food intake with intake frequency, which allows to evaluate human health risk with more care and to compare between different dietary patterns.

3) A qualitative research on the dynamics of diverse response to EDSs restriction among the textile manufacturers in southern Vietnam *(a societal reflection)*:

   Elaborated on the institutional theories, the study tests the influence of perceived adaptabilities, risks, benefits, and barriers on the textile firm’s attitude and behavior. It proposes a theoretical model of the textile firm decision-making, where perceived adaptabilities indirectly influence firm’s behavior through perceived technical risks, and market segment plays a moderating role in the patterns of risk and benefit perception. The findings are beneficial for regulatory agencies in designing policies for ecological and human health protection as well as reinforcing the textile exporting capacity of Vietnam.

4) An investigation of the distribution and removal of EDSs from the textile wastewater of a cotton and a synthetic fiber factory in southern Vietnam *(a physical reflection)*:

   The study provides evidence and a better understanding of EDSs removal efficacy as well as influencing factors in Vietnamese textile wastewater treatment. The results are beneficial for the textile industry in Vietnam regarding investment decisions for wastewater treatment.

1.1 Background

1.1.1 Endocrine disrupting contamination

There is a growing concern about the environmental contamination of endocrine disrupting compounds (EDCs) due to their adverse effect on wildlife and human despite of their low levels in the environment (Miodini et al. 1999, Rajapakse et al. 2002). An endocrine disruptor is defined as "*an exogenous substance or mixture that alters function(s) of the endocrine system and consequently causes adverse health effects in an intact organism, or its progeny, or (sub) populations*" ([IPCS] International Programme on Chemical Safety 2002). The endocrine system comprises of glands that secrete hormones and receptors for controlling developmental and various physiological processes in varied life stage of organisms (Melmed S. and Williams 2011). Five mechanisms that EDCs interfere with hormone signaling system have been identified: (1) mimicking normal hormones, (2) antagonizing hormones, (3) altering the synthesis and metabolism of hormones, (4) modifying hormone receptor levels; and (5) indirectly influencing via the immune and nervous systems (Devillers 2009). Hormone-driven-characterized stages such as utero, infancy, childhood, puberty, and menopause were particularly the most sensitive to exposure to endocrine disruptors (Muller 2013, Crain et al. 2008, Gore et al. 2014, Nepelska et al. 2014, Rachoń 2016). Evidence of endocrine disrupting effects in humans and wildlife has been found and reviewed in details by (WHO-UNEP 2013).

Endocrine disruptors are derived from natural and anthropogenic sources; nevertheless, global concern is related to the synthetic chemicals. Valuable reviews of Nassar et al. (2009), Casals-Casas and Desvergne (2011), Campbell et al. (2006) reveal major categories of EDCs. Prominent pesticide families known as EDCs include organochlorine pesticides (OCPs), organophosphates, carbamates, triazines, and pyrethroids. EDCs with industrial applications comprise alkylphenols (e.g., nonylphenol, octylphenol) and polyfluoroalkyl compounds (e.g., perfluorooctane sulphonate - PFOS, perfluorooctanonate - PFOA) that are widely used as surfactants, bisphenol A (BPA) and diethylhexyl phthalate (DEHP) as plasticizers, polybrominated diphenyl ethers (PBDEs) as flame retardants. Other categories are natural and synthetic hormones, pharmaceuticals and personal care products (PPCPs), and heavy metals (e.g., cadmium, lead, mercury) (Esplugas et al. 2007). The chemicals are released into the environment via various pathways such as sewage and industrial effluent discharge, run-off, emission and deposition.

The screening of 1153 organic micro-pollutants in the Vietnamese aquatic environment followed by a preliminary environmental risk assessment reveals the greatest risk quotient for nonylphenol (MEC/PNEC[1] = 128) (Chau et al. 2015). As such, nonylphenol is indicated as one of the substances of high concern and warrant further studies. The chemical and its ethoxylates are surfactants known to be responsible for endocrine disrupting effects.

1.1.2 Endocrine disrupting surfactants

Nonylphenol (NP) and its ethoxylates (NPEOs) are widely used as surfactants in numerous industries such as cleansing, textile, personal care products, pesticides, paper, and plastic, and end up in the environment via municipal and industrial discharge ([RPA] Risk & Policy Analysts Limited and [BRE] BRE Environment 2003). Industrial and institutional cleaning sector appears as the greatest consumer of NP/NPEOs, accounting for 30% of the total production in Europe (RPA 1999) and 14% in Japan ([NITE] National Institute of Technology and Evaluation 2003). The second large consumer is the textile industry with a share of 10% and 13%, respectively (RPA 1999, NITE 2003). Additionally, NP/NPEOs are widely used as surfactants in cosmetics, personal care products, and pharmaceuticals. The chemicals end up into the environment as sewage and industrial effluent discharge, as well as sludge disposal on soil. NPEOs and particularly NP are responsible for endocrine disruption among wildlife and human beings (Ejlertsson et al. 1999, Ahel et al. 1994a); therefore, the chemicals are hereafter referred to as endocrine disrupting surfactants (EDSs).

NP is an input in the production of NPEOs, and vice versa, NPEOs undergo several processes of decomposition to formulate short-chain homologues, nonylphenol ethoxycarboxylates (NPECs), and NP as a final product. Although photochemical degradation of NPEOs and NP has been recognized (Ahel et al. 1994d), biotransformation remains as a predominant process (Ahel et al. 1994c, Mann and Boddy 2000, Manzano et al. 1999, Potter et al. 1999, Scott and Jones 2000). Under aerobic conditions, NPEOs may undergo a complete oxidation producing short-chain homologues and NPECs (Maki et al. 1994, Di Corcia et al. 1998). NP is formulated from NPEOs particularly under anaerobic or reducing conditions (Ahel et al. 1994a, Charles et al. 1996). The mineralization of NP, however, is eager to occur under aerobic, particularly nitrifying conditions (Birkett and Lester 2003, Cirja et al. 2008, Loyo-Rosales et al. 2007b, Scruggs et al. 2004). It is suggested that NP seems persistent under reduced and anaerobic conditions (De Weert et al. 2010, De Weert et al. 2009, Shang et al. 1999, Ejlertsson et al. 1999). The biotransformation of NPEOs and NP is illustrated in Figure 1.1.

[1] MEC: Measured Environmental Concentration; PNEC: Predicted No-Effect Concentration; MEC/PNEC ratio of greater than 1 indicates a potential hazard or risk.

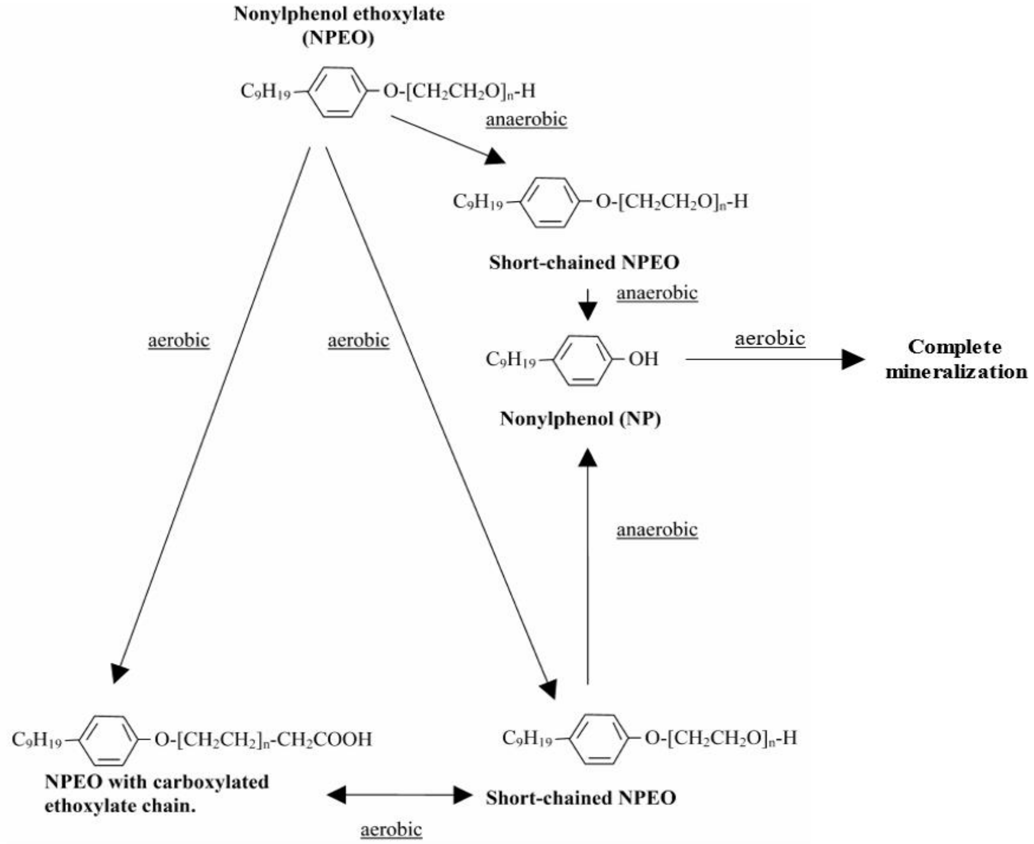

Figure 1.1 Biodegradation pathways for NPEOs and NP in the environment
Source: modified from Sarmah and Halling-Sørensen (2007).

NP shows to be persistent at different levels in the environment (Lopez-Espinosa et al. 2009, Manzano et al. 1999, Ejlertsson et al. 1999). On the other hand, NP has a high affinity for solids such as sediment, sewage sludge, and soil amended with sewage sludge (Ahel et al. 1994a, Ahel et al. 1994b), as well as for lipids (Ademollo et al. 2008), hence it has been shown to accumulate in organisms (Ahel et al. 1993, Ying 2006).

Past research has demonstrated the presence of NP in all trophic levels such as plankton, benthic invertebrates, fish, birds, and mammals (Casatta et al. 2015, Casatta et al. 2016, Diehl et al. 2012, Gu et al. 2016, Hu et al. 2005, Isobe et al. 2007, Korsman et al. 2015), via a complex food web (Kidd et al. 2012). NP has been detected in foodstuffs (Chen et al. 2010, Gu et al. 2016, Guenther et al. 2002, Gyllenhammar et al. 2012, Lu et al. 2007), drinking water (Shao et al. 2005), human adipose tissue, urine, maternal blood plasma and amniotic fluid, blood serum, and breast milk (Ademollo et al. 2008, Azzouz et al. 2016, Chen et al. 2010). The presence of NP in pregnant women's decidua and early embryos along with maternal transfers has also been observed (Chen et al. 2016).

NP has raised concern due to its endocrine disrupting effects on wildlife and humans during the past two decades. Adverse effects of NP on reproductive, immune, and central nervous systems have been discovered in fish, rats, birds, and humans with possible abnormalities in embryos and offspring (Cosnefroy et al. 2009, Ghisari and Bonefeld-Jorgensen 2005, Jie et al. 2010, Jobling et al. 1996, Kim et al. 2006, Mao et al. 2010, Nakazawa and Ohno 2001, Pedersen et al. 1999, Razia et al. 2006, Soto et al. 1991, Vosges et al. 2012, Couderc et al. 2014, [EC] European Commission 2016). Recent studies on carcinogenesis have reflected the relation of exposure to NP to the possibilities of breast cancer in women (Wu et al. 2008) and prostate cancer in men (Forte et al. 2016, Kim et al. 2016). A study of Lepretti et al. (2015) revealed negative impacts on human intestinal homeostasis and functionality. The mechanisms of action of NP are related to xenoestrogens (Pillon et al. 2005, Mnif et al. 2007, Kuiper et al. 1998), antiestrogens (Preuss et al. 2010), and disruption of thyroid function (Hofmann et al. 2009), which occur on nuclear (genomic) (Wu et al. 2008), extranuclear (non-genomic) (Thomas and Dong 2006), and cross-talk between genomic and non-genomic pathways (Li et al. 2006, Bulayeva and Watson 2004).

Among the surfactants, NP is the most ubiquitous substance which accounts for about 80% of alkylphenols in the surface water in Vietnam (Tri et al. 2016). NP has been detected at extremely high levels in urban watercourses such as in the cities of Ha Noi and Ho Chi Minh, in the range of 0.02 – 9.7 μg/L (mean 3.0 μg/L) and 2.0 – 20.0 μg/L (mean 9.7 μg/L), respectively (Hanh et al. 2014). The water samplings in Red river estuary (Red river delta), Haiphong habour (Haiphong city), Halong bay (Quangninh province), Balat estuary (Thaibinh province), Huong river estuary (Hue city), Longxuyen city (Angiang province, Mekong delta) have provided evidence that NP is ubiquitously distributed in the urban areas of Vietnam (Viet et al. 2006, Duong et al. 2010, Tri et al. 2016). The concentration of NP in the leachate from the northern landfills ranges from 46 to 2,030 ng/L (Tri et al. 2016), suggesting the existence of NP and its ethoxylates in varied domestic products. Regarding environmental risk, it is suggested that NP may cause ecological effects due to its high risk quotient (Chau et al. 2015).

1.1.3 Risk management of endocrine disruptors

Concerning the negative impacts on the ecosystem and the health of human beings, NP/NPEOs have been added to the list of chemicals for priority action for the protection of the marine environment of the North-East Atlantic since 1998 by the OSPAR Commission (Slovic et al. 1991a). Since 2000, NP has been classified as a priority hazardous substance under the Directive 2000/60/EC of the European Parliament and the Council, followed by the restrictions in marketing and use of NP and its ethoxylates under the Directive 2003/53/EC. Three years later, the chemicals have been regulated under the Registration, Evaluation, Authorization and Restriction of Chemicals (REACH) program of the European Commission in 2006 ([EC] European Commission 2006). Under this scheme, the residual levels of the chemicals of equal or greater than 0.01% by weight in textile articles are not allowed to be placed on the European market after 3 February 2021 (EC 2016). In 2008, the concentration of NP in surface waters has been regulated in the Directive 2008/105/EC. Accordingly, the annual average level of NP should not exceed 0.3 μg/L. Other countries such as Taiwan, Korea, and the United States have also launched or proposed safety requirements regarding the residues of NP/NPEOs in textile products with a focus on infants and children health protection ([AAFA] American Apparel and Footwear Association 2017,

[BSMI] Taiwan Bureau of Standards Metrology and Inspection 2013, [KATS] Korean Agency for Technology and Standards 2016).

Nevertheless, NP and NPEOs are allowable for use today in many Asian countries including Vietnam. None of the current regulations specifies NP/NPEOs but regulates the substances under general indicators such as "surfactants" in domestic and textile wastewater, or "phenolic compounds" in drinking water and general industrial wastewater, or both types in surface water quality ([MONRE] Ministry of Natural Resource and Environment 2008, [MOH] Ministry of Health 2009, [MONRE] Ministry of Natural Resource and Environment 2015, 2011a, b). Since the design of water/wastewater treatment facilities and the monitoring of water/wastewater quality solely rely on these regulations, the control of the environmental contamination of NP/NPEOs is not guaranteed by the existing regulatory system.

1.1.4 Knowledge gaps in EDCs risk management

At this moment, potential risks posed by EDCs are not yet fully understood. Fuhrman et al. (2015) provide a comprehensive discussion on the unknowns and scientific uncertainties for EDCs which could be challenges for traditional risk assessment. Elaborating on the material, three knowledge gaps has been identified to be of interest to this research.

First, *timing of exposure* is suggested to be as important as dose for having endocrine disrupting effects on organisms (Fuhrman et al. 2015). Exposure durations (e.g., short/ long period of time, lifetime) and particularly exposure frequency (e.g., intermittent/ regular basis) are two factors that influence toxicity (USEPA 2000). Current method of health risk evaluation based on chronic effect test endpoints and regular exposure may lead to an overestimation of health risk. Therefore, the effect of *intermittent* or *less regular exposure* should be realized in health risk evaluation.

Second, some EDCs have been found to display potential *trans-generational effects* due to their persistent characteristic, and nonylphenol is among those substances (Fuhrman et al. 2015). This elucidates the role of parents towards their next generations. It is supposed that people with awareness about EDCs and potential health risks are more prepared for reducing or avoiding adverse effects on their children. Therefore, the insights of *societal awareness* and *risk perception* regarding EDCs are essential for designing appropriate social policies.

Third, due to unknowns and uncertainties it is suggested that risk communication needs to *balance precaution and alarm* (Fuhrman et al. 2015). Exposure via ingestion route is related to a variety of food products, in which some could be consumed with precaution whilst others need health risk warning. This emphasizes the need of *the insight on contamination for single food products*.

1.2 Theoretical framework

1.2.1 Multi-perspective of risk

Science and technological innovations (e.g., in chemical industry, nuclear energy, nano-materials) have facilitated human beings in the world with more convenience, comfortable, and prolonged life; nevertheless, they also pose risks to ecological and human well-being. As such, systematic and technically risk assessment methods are emphasized as a basic means of approach to environmental management decisions in modern societies (Duah 1998). Risk appraisal for an environmental contamination problem attempts to examine the contamination from various aspects: the nature, the sources, the extent of influence, the populations at risk, the most significant exposure pathways, the likelihood of health and environmental effects, and the tolerable or acceptable level, which is useful for deriving corrective actions, preventive measures, and relevant policies (Duah 1998). For those purposes, risk assessment is designated as a core content in any ecological and health risk management scheme.

Whist the roles of risk analysis are well illustrated in the environmental and health protection relevant to numerous domains, such as hazardous chemicals and oil spills, radioactive and hazardous waste disposal, and EDCs, it is interesting to look back to some well-known events in the United Kingdom, the United States, and Japan where risk-analysis-based decisions result in unsatisfied consequences. The first case is about Shell cooperation's proposal of burying the waste oil (Brent Spar) in the North Sea in the late 20th century. After a large study on technical, safety, and environmental aspects to prove it as the best practicable option, Shell's proposal has been approved by the UK Department of Trade and Industry in 1994 (Cowell et al. 2002). However, this decision later turns into abandon since it faces serious campaign of boycott from interest groups and public whose roles were not taken into account in the decision-making process (Löfstedt and Renn 1997). The second case portraits the distrust of Nevada residents towards the US Department of Energy (DoE) and the advantages of technologies in risk management of nuclear waste. The distrust is reasoned for the deteriorated reputation in the DoE's risk management capacity due to their failures in radioactive waste management at the nation's military weapons facilities and the leakage of unprecedented radiation from Hanford weapons plant (Washington) in the 1940s and 1950s (Slovic et al. 1991b). Consequently, this leads to a dramatic frustration, disturbance, and political opposition to the project on nuclear fuel waste disposal in the US. The third case, the accident of Fukushima Dai-ichi nuclear plant explosion in Japan, gives an example of the failure in nuclear risk governance. The policy of nuclear energy promotion is based on an introverting attitude coupled with perceived benefit and underestimated perceived risk in the context where there being deficiencies in risk-related knowledge and in organizational capacity for managing risks and emergency response among different institutional levels (Taniguchi and Shiroyama 2017). After the explosion, the situation is even worse since the government as well as the nuclear energy industry are faced with a significant anxiety in a wider population and unexpected responses to fear, particularly with regards to so-called "lower - risk" cases (Robertson and Pengilley 2012). Among the causes of this are supposed to be "scientific uncertainties about low dose radiation" and "the ambiguous effects of low dose radiation to the environment" (Tateno and Yokoyama 2013). The stories have provided evidence that inappropriate top-down decisions that merely rely on risk analysis and without considering public perception have caused the distrust in industries and governmental authorities, leading to failures in risk communication and environmental decision-making. This reveals the substantial roles of stakeholders and public participation relative to the roles of scientists in risk decision-making processes, particularly when scientific uncertainties are high.

It is claimed that "there is no such thing as real risk" (Slovic 1992), and "danger is real, but risk is socially constructed" (Slovic 1999). From socio-cultural perspective, risk judgments by both scientists and public are influenced by gender, race, worldview, ideologies, values, emotion, affect, and trust in different ways (Buss et al. 1986, Mary Ann and Witt 1989, Slovic et al. 1991a, Dake 1992, 1991, Slovic and Peters 1998, Slovic 1997, Slovic et al. 1997, Finucane et al. 2000, De Groot et al. 2013). From psychological perspective, public view risk through colorful lenses based on their experience and feelings of "uncertainty, dread, catastrophic potential, controllability, equity, risk to future generations and so forth" (Slovic 1999). Meanwhile, scientists define risk as probability-based risk based on statistical data, direct experience, models, and approximation with probably subjective judgments and assumptions (Slovic 1997, Aven 2008, Aven and Renn 2010). Hence, risk analysis by scientists should not be able to claim universal values rather than stakeholders' and lay people's judgments, and it exists both societal and physical reflections of risk instead (Figure 1.2) (Aven and Renn 2010). Controversy in risk perception towards a so-called hazard has been reported in different scales, not only individuals and groups as mentioned above, but also beyond geological boundaries (nations). One example of this is given by Slachtová et al. (2003) whose investigation in the public of two adjacent countries results in opposing views of environmental health priorities. In short, societal perspectives of risk are diverse and may not hold true for traditional analyses of risk. Hence, both technical and societal perspectives of risk are able to contribute equally to an innovative risk decision-making process.

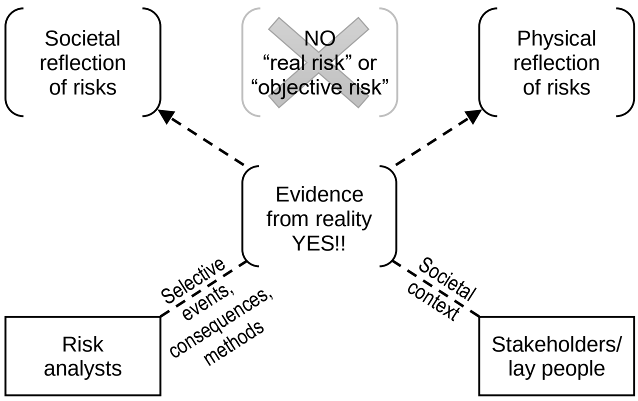

Figure 1.2 Reflections of risk from risk scientists and non-scientist people

1.2.2 A perception – response – evaluation (PRAVE) framework for EDSs risk decision-making

From the illustrative stories and risk theories, the author propose that *risk analyses* conducted by scientists and studies of *risk perception and response* among stakeholders and public should be of equal importance and complement environmental risk management. The integration provides useful information for designing risk communication strategies, formulating more comprehensive measures, and providing additional knowledge and normative criteria to evaluate them (Fischhoff et al. 1984). On the other hand, it strengthens the legitimacy of risk process (Slovic 1999, Fuhrman et al. 2015). This motivates the formulation of an innovative framework for EDSs risk management. The framework involves both types of knowledge, which serve as equally valuable and complementary inputs to the EDSs risk communication at local level and decision-making at administrational level: 1) Risk Perception and Response (societal reflections of risk), and 2) Risk Evaluation (physical reflections of risk). It is hereinafter referred to as PRAVE framework. The following sections explain how the theoretical framework is formulated and its structure.

Risk perception and response

- *Public theories on risk perception and response*

It is aforementioned that EDSs may cause adverse effects on human communities who are exposed to the chemicals. Under a given environmental situation, the level of impact may contribute to how responses to EDSs risk are taken by the exposed communities. For this reason, the insights of people behavior and their decision-making mechanisms are beneficial for designing strategic response to the risk of EDCs in Vietnam. Two types of behavior: diet-related and non-diet-related are of concern.

In risk assessment process, understanding risk perception plays a key role in explaining risk acceptability and predicting risk response (Slovic 1987). Theories of risk perception are initially developed on the basis of value-belief-norm (VBN) theory (Stern 2000, Stern et al. 1999) which intends to explain pro-environmental behavior. In expanded literature, the relationship of risk perception - risk acceptability predicting environmental action has been demonstrated in several fields, such as nuclear energy (De Groot and Steg 2010, De Groot et al. 2013, Peters and Slovic 1996), nuclear waste (Sjöberg and Drottz-Sjöberg 2001), flood risk due to climate change (De Dominicis et al. 2015), environmental hazard (Lindell and Hwang 2008), and ecological risk (Slimak and Dietz 2006). Risk perception terminology captures one's belief in negative consequences of an event as true (risk belief) and one's concern about the consequences (risk concern) (Bish et al. 2011). The existing theories of risk perception could also be useful to explain people's self-protective response to the health risk related to EDCs.

Perceived risk depends on individual capacities such as knowledge (Agyei-Mensah and Oteng-Ababio 2012, Gaash et al. 2003, Kung and Lee 2006), insight knowledge (Che et al. 2014), prior personal experience (Lindell and Hwang 2008), and self-controllability (McDaniels et al. 1997). Among few researchers who study EDCs risk perception, Maxim et al. (2013) indicate that although people have rather good knowledge on a particular issue, they may feel uncertain about that issue because they are not able to link the knowledge with the surrounding situation. However, the roles of knowledge and perceived uncertainty in explaining EDCs risk perception have not been well established due to a lack of qualitative research.

Elaborating on the existing literature it is supposed that people's self-protective actions depend on their perception of EDSs risk; whereas risk perception is governed by the knowledge of the general situation of environmental pollution and the knowledge of EDSs as well as the perception of uncertainty regarding the knowledge. Hence, *knowledge of different levels*, *perceived uncertainty*, *risk perception*, and *self-protective response* are important factors in this framework. Since public expectation to the government for EDSs risk control is of concern, *risk acceptability* is also taken into account in this framework.

- *Firm behavior theories*

EDSs risk is also related to the sources of EDCs (mainly as industrial consumption and discharge). Industrial firms' response, such as material substitution, manufacturing improvement, and wastewater treatment, also influences the extent to which EDSs present in the environment. Therefore, the framework also includes the knowledge of industrial institutions as valuable inputs in the EDSs risk management scheme.

The existing literature reports the influences of external factors such as legal, social, and economic pressures (Hoffman 2001, Gunningham et al. 2004, 2003, Liu et al. 2010, Singh et al. 2014), and internal factors such as resources and strategies (Aragón-Correa et al. 2008, Christmann and Taylor 2006, Galende and de la Fuente 2003) on firm behavior and performance. The effects on firms may differ depending on firm characteristics, organizational structure, and learning experience (Delmas and Toffel 2004, Meeus and Oerlemans 2000, Passetti and Tenucci 2016). However, it has been argued that individual perspectives such as managerial attitude, value, and the perceptions of costs, risks, and benefits associated with certain behaviors play salient roles in driving firm managers to take different actions (Sharma 2000, Egri and Herman 2000, Wu and Wirkkala 2009, Ervin et al. 2013). This is because firm managers interpret external pressures differently depending on parent company and plant characteristics (Delmas and Toffel 2004). In addition, it has been recognized that organizational and technological adaptabilities also influence firm innovative behavior and performance (Meeus and Oerlemans 2000). Elaborating on the relevant theories, this framework includes an investigation of *firm behavior* from the aspects of individual's *awareness*, *perception*, and *attitude*, where perception may be relevant to adaptabilities, risks, benefits, and barriers.

Risk evaluation

Physical investigations of risk serve as basic contents in any risk decision-making scheme. The aim is to answer basic risk-relevant questions about the nature, the sources, the extent of influence, the populations at risk, exposure pathways, and the likelihood of health and environmental effects (Duah 1998). As such, a risk assessment comprises of four elements: 1) Hazard identification (e.g., contaminants of concern), 2) Exposure assessment (e.g., environmental fate and transport processes, estimation of exposure point concentrations, and quantification of exposures), 3) toxicity assessment (e.g., dose-response assessment), and 4) risk characterization (e.g., ecological and human health effect) (Duah 1998, [WHO] World Health Organization 2002). Risk evaluation of an identified contaminant is based on the outputs of exposure and toxicological assessments.

Hazard identification involves gathering and evaluating data on health effects and the pathways of exposure, and identifying the sources of specific contaminants of concern (Duah 1998). This phase is relevant to both qualitative and quantitative methods of data generation.

Quantitative methods useful for exposure assessment include field measurements, and predicting models (e.g., multimedia fate – transport models, toxicokinetic models). Multimedia fate – transport models predict the dispersion, degradation, and concentration of contaminants in different environmental compartments. Valuable reviews of fate models are provided by Park et al. (2008), Suter II (2016). Toxicokinetic models are used for predicting bioaccumulation of contaminants in biota. Useful information of eco-toxicological models could be found in Barber (2003), Galic et al. (2010), Fath et al. (2011), Park et al. (2008), Arnot and Gobas (2004), Arnot et al. (2008), Hendriks A. J. and Heikens A. (2001), Hendriks A. J. et al. (2001).

Health risk evaluation is based on qualitative methods of comparing receptor chemical intakes with doses that are shown without adverse effects. Traditionally, the "tolerable daily intake" (TDI - mg/kg body weight. day), "reference dose" (RfD – mg/kg body weight. day), and "reference concentration" (RfC – mg/m^3 or µg/L) are often used for noncarcinogenic effects; whereas the "slope factor" (SF – $[mg/kg.\ day]^{-1}$) and unit risk factor (URF – $[\mu g/m^3]^{-1}$) are used for carcinogenic effects (Duah 1998). The TDI is the amount of a substance to which a receptor can be exposed on a daily basis over an extended period of time without suffering a harmful effect. The RfD and RfC refer to the maximum amount of a chemical that humans can absorb on a regular basis without experiencing chronic health effect. The RfDs and SFs are used in evaluating risk from oral or dermal exposures, whilst the RfCs and URFs are for inhalation exposures.

Giving the latest toxicological knowledge on EDSs, *hazard identification,* and *exposure and health risk characterization* present as two major physical domains in this framework. In the view of this study, a traditional hazard identification may not be sufficient for screening local contamination of EDSs since this is related to diverse sources of discharge. Instead, an *advanced hazard identification* based on field measurement and analytical assessment is needed to strengthen the knowledge of EDSs sources. This can be obtained by studying the distribution and removal efficiency of EDSs from wastewater treatment facilities.

Based on the theoretical framework, four studies will be conducted with two aims: 1) to fulfill the knowledge gaps that have been identified in Section 1.1.4; 2) to achieve insights for EDSs risk decision-making. The studies cover the following topics: 1) societal perspectives and self-protective behavior (study 1); 2) ecological and human health risk (study 2); 3) firm's perspectives and response (study 3); and 4) distribution and removal of EDSs (study 4).

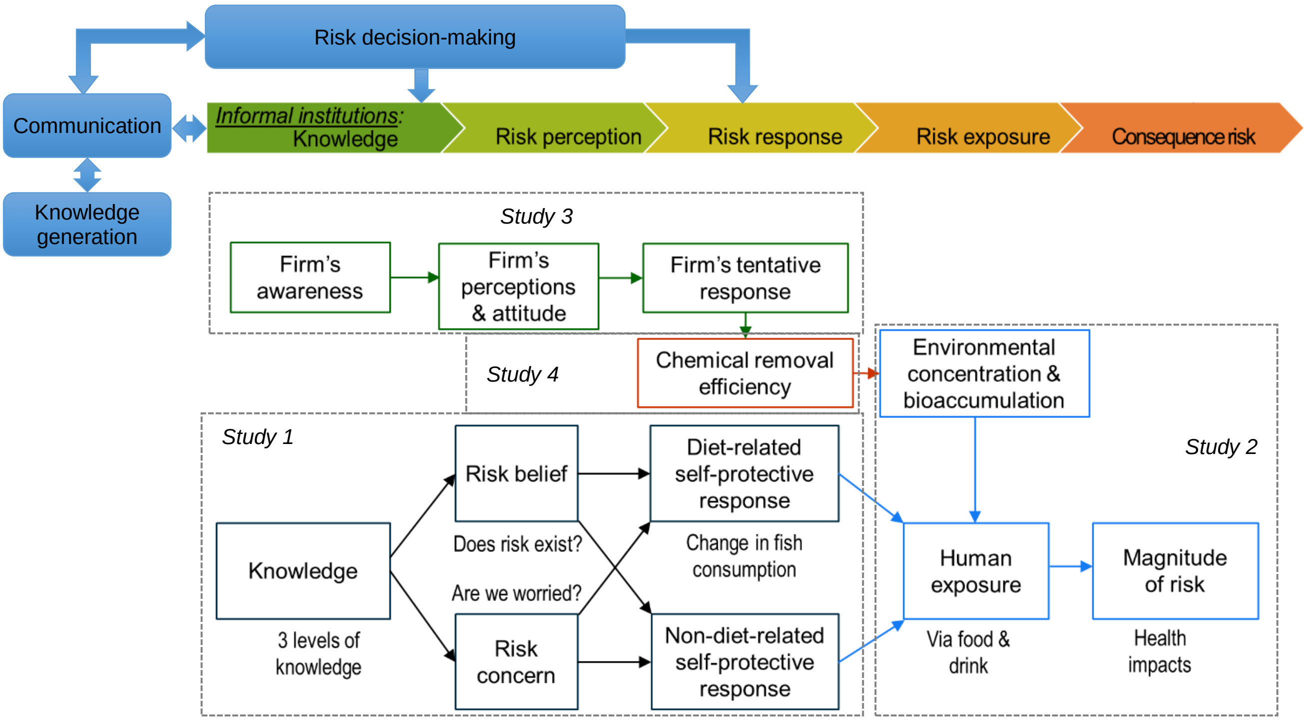

Figure 1.3 The PRAVE framework for EDSs risk decision-making

1.2.3 Mechanism of PRAVE

PRAVE could have positive impact at society level. It highlights the role of risk response in enhancing or reducing environmental contamination and risk exposure with regards to EDSs. In this manner, for example, people adopt self-protective response so that they are less likely being exposed to risk, whilst firms which take pro-actions could help improve the environmental situation. The knowledge of public self-protective response and firm behavior sheds light on the exposure to EDSs risk, as well as supports the prediction of ecological risk. The insights are subsequently subjected to EDSs risk communication and decision-making. Outcome could be educational programs that aim to impact public behaviors through strengthening their awareness of EDSs and risk perception. Additionally, enforced regulations and incentives could be solutions to direct firm behavior and thus indirectly tackle the contamination problem.

1.3 Motivation, research questions, and objectives

Acknowledging the scientific uncertainty on the causal-effect concerning NP exposure in humans, regulations on chemical safety, restriction of marketing and use, environmental quality standard, and residue control have been promulgated with the aim of ecological and health protection within the European Union. Asian countries such as Taiwan, Korea, and Japan have also launched monitoring programs as well as proposed limiting NP in textile articles, especially for children. Nevertheless, although the concentration of NP in Vietnamese urban waters has reached alarming levels in recent years (Duong et al. 2010, Hanh et al. 2014, Minh et al. 2016, Viet et al. 2006), the use of this chemical is still permitted and the response to ecological and health risk is negligible. This raises two research questions: *(1) how the society perceives the risk of these endocrine disrupting surfactants (risk perception),* and *(2) how people expect to avoid the risk (risk acceptability and risk response)*. Answering these questions is practically vital for designing strategic response to the risk of EDCs in Vietnam. Thereby, the first study was conducted with the objective:

> ***Objective 1:*** To examine the roles of the awareness of different levels and perceived uncertainty on EDSs risk perception, acceptability, and self-protective response among public using structural equation modeling (SEM).

Recent studies have demonstrated the distribution of NP in the waters across Vietnam and the accumulation of NP in lower waters that receive wastewater from populous urban areas (Viet et al. 2006, Hanh et al. 2014, Duong et al. 2010, Tam et al. 2016). It is indicated that the municipal wastewater of Ho Chi Minh city has deteriorated the aquatic environment of the Can Giuoc river, which flows down through the territory of Long An province (Ha and Phep 2011, Duc et al. 2016). The river serves as the main water source for aquafarming activities and the supply of natural fishery products for the local people in the Can Giuoc – Can Duoc basin. Hence, exposure to NP via riverine food chains may pose a health risk on wildlife and the local people. It is essential for risk scientists and governmental authorities to know: *(1) how NP is distributed in the Can Giuoc river*, *(2) to which levels the local aquatic ecosystem and the local people are exposed to NP*, and *(3) how the health risk from riverine food consumption is better characterized for the purpose of risk communication*. This motivated the second study with the objective:

> ***Objective 2:*** to develop an integrated modelling framework for evaluating bioaccumulation and health risk from riverine food consumption.

The endocrine disrupting surfactants NP and its ethoxylates have been detected in textile products in many countries of manufacture and purchase, which provides evidence that the chemicals are used within transnational clothing supply chains (Brigden et al. 2012b, Brigden et al. 2013a, Brigden et al. 2012a). The chemicals are released into the environment during manufacturing processes and use of textile products, mainly as wastewater discharge (Loos et al. 2007, Pothitou and Voutsa 2008, Brigden et al. 2011, Brigden et al. 2013b, Ho and Watanabe 2017). Concerning the adverse effects on wildlife and humans, many countries have launched ecological and health protection policies including regulating the residual levels of NP/NPEOs in the imported textile articles. As such, the use of NP/NPEOs as surfactants in the textile industry in Vietnam not only poses local ecological and human health risks but also emerges as a barrier for exporting textile goods to foreign markets, such as European countries and America, due to the regulations on the residual levels in textile products. Therefore, eliminating the use and discharge of NP/NPEOs is crucial. This arouses a question as to *how the textile manufacturers in Vietnam response to the urgency of NP/NPEOs restriction in their manufacturing processes and products*. The insight of textile firm's behavior is beneficial for regulatory agencies in designing appropriate policies for ecological and human health protection as well as reinforcing the textile exporting capacity of Vietnam.

Whilst various types of enterprise risk have been recognized (Kenett and Raanan 2011) and firm's responses are diverse, existing literature is insufficient to explain the dynamics of firm decision-making. Therefore, a theoretical framework that helps gain more insight into the mechanism of firm's response is needed. The third study is conducted to fulfil this gap.

> ***Objective 3:*** To propose a firm behavior model based on a qualitative analysis of the Vietnamese textile firm's response to the restriction of nonylphenol and nonylphenol ethoxylates through attitude and the perception of adaptabilities, risks, benefits, and barriers.

In textile industry, NPEOs are widely used as detergents and auxiliaries (Antal et al. 2016) in various processes, such as wool scouring, hydrogen peroxide bleaching (Arnot et al. 2008), washing, dyeing, and printing (Cobbing et al. 2013, Hanh 2015). Treatment of textile wastewater before discharge is mandatory in Vietnam. However, the designs of wastewater treatment facilities are mainly for macro-pollutant removal with a major application of conventional processes ([VEA] Vietnam Environment Administration 2011). Nevertheless, it is suggested that conventional processes are inferior to effectively remove NPEOs and NP in wastewater (Brigden et al. 2013b, Zhou et al. 2009). Therefore, investigations on the existing textile wastewater treatment facilities in Vietnam regarding NP/NPEOs removal are needed. This motivated a field investigation study with the objective:

> ***Objective 4:*** To investigate the distribution and removal of NP/NPEOs across two typical textile wastewater treatment processes and to assess the influence of hydraulic retention time (HRT), nitrifying conditions, solids retention time (SRT), mixed liquor suspended solids (MLSS), and water temperature on NP/NPEOs removal.

Operationalization of the objectives 1 – 4 are accordingly articulated in studies 1 – 4.

1.4 Scope of research

Recent studies reveal alarming levels of NP in the urban surface water and sediment across Vietnam (Hanh 2015, Chau et al. 2015, Duong et al. 2010, Viet et al. 2006, Tri et al. 2016). It is suggested being contributed to domestic activities that link with the use of cleaning products, personal care products, and cosmetics. Therefore, this research places its focus on EDSs contamination in the urban cities in Vietnam. Ho Chi Minh city, Da Nang city, and Long An province are the main places in Vietnam where the research is conducted (see Figure 1.4).

Studies of Chen et al. (2010), Brigden et al. (2013a), Axelrod (2015) have demonstrated the worldwide presence of NPEOs in the majority of textile products regarding all materials and across most of the countries of manufacture and consumption. The authors suggest that lower traces of NPEOs in the textile products imply a higher discharge of NPEOs into the watercourse during manufacturing processes. Recent investigations in textile manufacture in developing countries such as Thailand, China, Mexico and Indonesia by Greenpeace have revealed that NP/NPEOs are among the most commonly detected hazardous chemicals in the effluent of wastewater treatment plants (WWTPs) (Cobbing et al. 2013).

On the other hand, Vietnam is one of the top ten textile exporters in the world. From 2004 to 2014, the textile industry achieved a compound annual growth rate (CAGR) of about 19% per year, increasing its contribution to Vietnam's GDP by 5% to 15% (Nguyet 2015). Textile industry is one of the most polluting industries releasing highly toxic and persistent chemicals into the environment, especially the watercourse (Hasanbeigi and Price 2015, Loan 2011). Therefore, this research places an interest on textile industry with regards to the use and removal of NP/NPEOs. The textile factories to be investigated in this research are located in Tay Ninh, Binh Duong, Long An provinces and Ho Chi Minh city (see Figure 1.4).

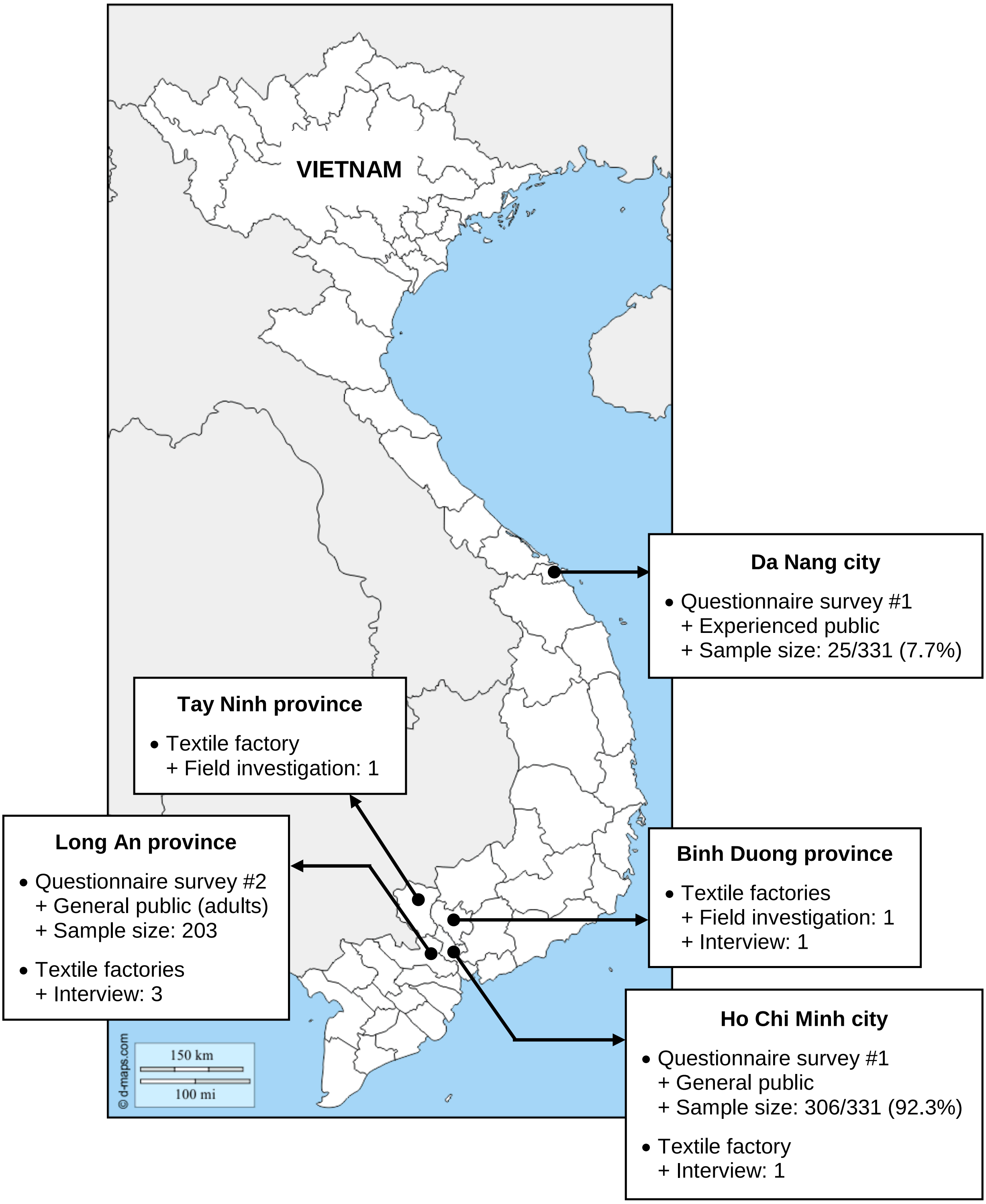

Figure 1.4 Study locations, types of survey and characteristics

(Source of original map: https://d-maps.com)

1.5 Structure of the dissertation

The contents of this dissertation are organized in seven chapters (see Figure 1.5).

Chapter 1 starts with a brief introduction on the studies that comprise the main components of this dissertation and their contributions. The following contents tell the context where this study is initiated through the background, the theoretical framework showing how the author views the problem to propose approaches to problem solving, the motivations, research questions, and objectives.

Chapter 2 presents the methodological framework of this research as a whole, as well as the description of the methodological framework for each study.

Chapters 3 – 6 present the individual studies that have been conducted in order to fulfill the objectives stated in Chapter 1.

Chapter 7 concludes the major findings as well as the implications of the dissertation.

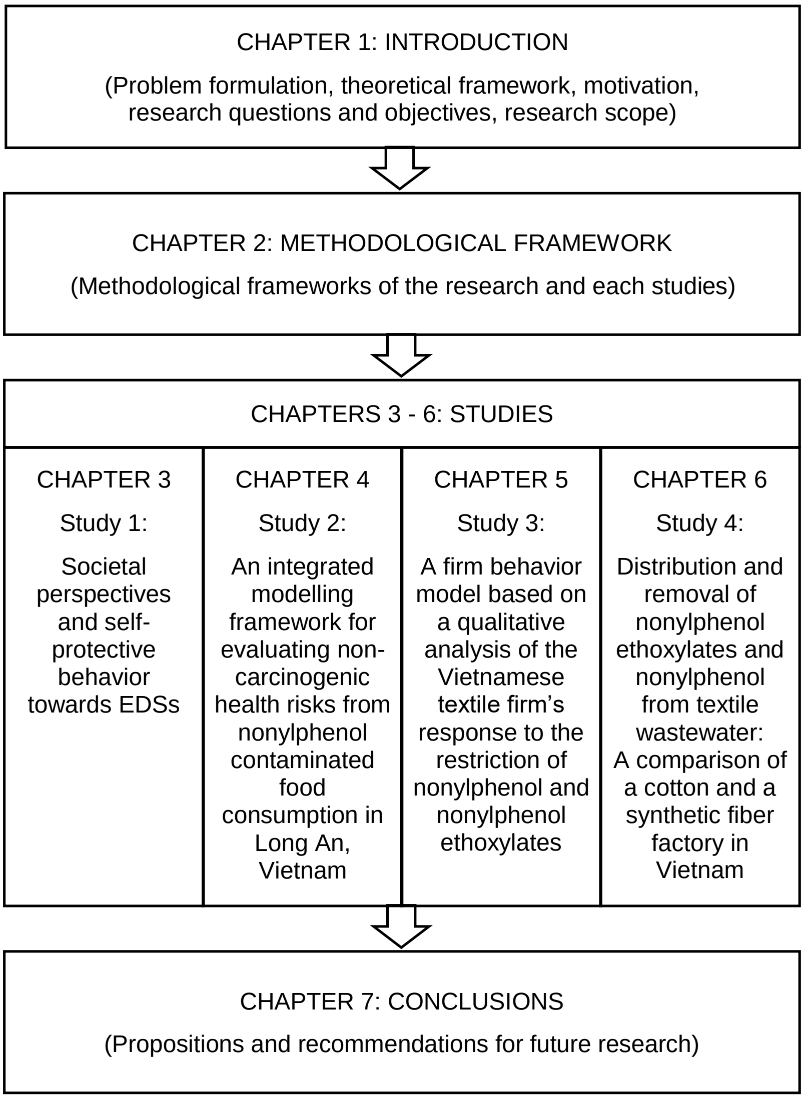

Figure 1.5 Structure of the dissertation

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and commission regulation (ec) no 1488/94 as well as council directive 76/769/eec and commission directives 91/155/eec, 93/67/eec, 93/105/ec and 2000/21/ec Brussels, Belgium.
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CHAPTER 2

METHODOLOGICAL FRAMEWORK

This research comprises both types of quantitative and qualitative approaches. The quantitative methods include data generation by means of questionnaire surveys and field measurements, which are subjected to quantitative data analyses with statistical modelling (study 1), chemical fate modelling, food web bioaccumulation modelling, exposure and health risk evaluation (study 2), and an analytical assessment (study 4). Another source of quantitative data of study 2 was literature-derived. Study 3 involves qualitative methods of data collection via semi-structured in-depth interviews and data interpretation by cognitive mapping and analytical assessment. Figure 2.1 presents an overview of the spheres covered in the four studies and their methods. The methodological framework of each study is described in the following sections.

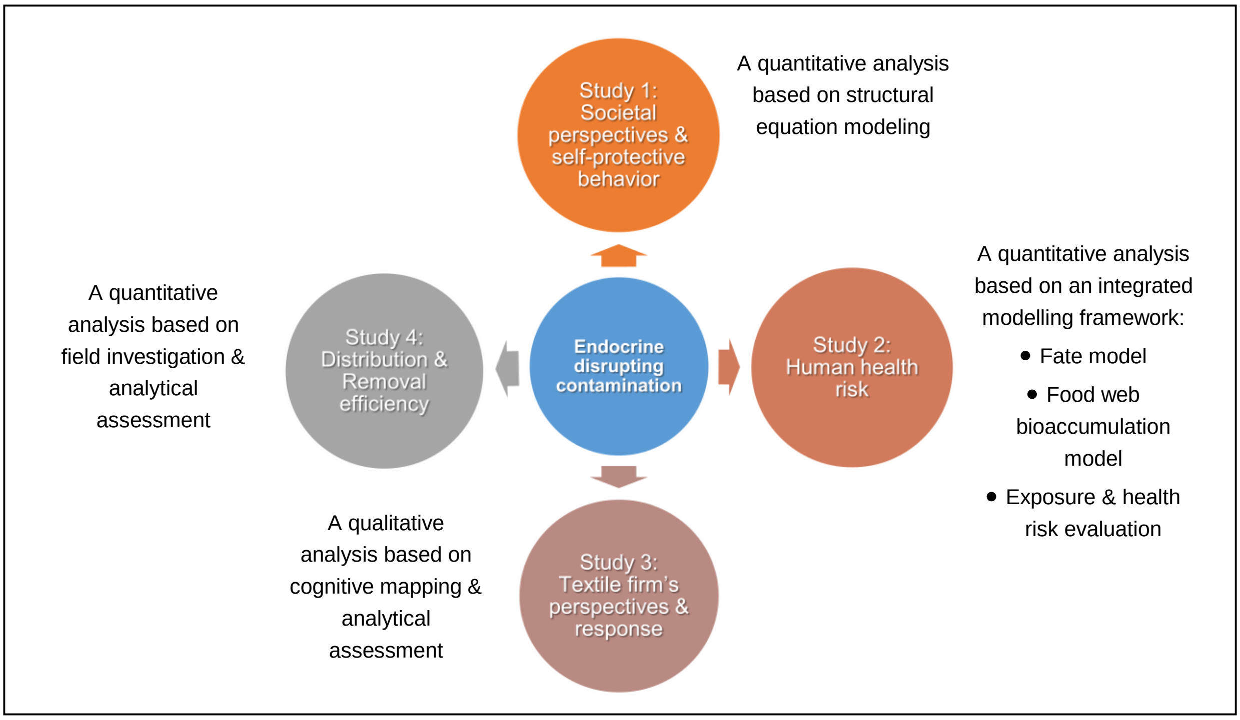

Figure 2.1 Methodological framework of the research

2.1 Study 1: Societal perspectives and self-protective behavior towards EDSs

Elaborating on existing theories, this study hypothesized that perceived uncertainty might play a mediating role in the relationship of knowledge of EDSs with risk perception (in terms of risk belief and risk concern), as well as in the relationships of risk perception with risk acceptability and resulting self-protective response. The hypotheses were detailed as follows:

Hypothesis 1 (H1): Perceived uncertainty mediates the positive relationships of specific awareness of EDSs with risk belief, risk concern, risk acceptability, and self-protective response.

Hypothesis 2 (H2): General awareness of water pollution and RHPs also has direct and positive effects on risk belief and risk concern.

Hypothesis 3 (H3): Perceived uncertainty plays a mediating role in the positive relationship of risk belief and risk concern with risk acceptability and self-protective response.

Hypothesis 4 (H4): People distinguished by experience and the status of pregnancy and child differ in the relationships of risk perception with risk acceptability and self-protective response.

Data for this research was obtained from a questionnaire survey among 331 adult participants of all genders, including pregnant women and young mothers, who were selected randomly. Most of the participants (92% of the sample) were recruited in sub-urban and urban Ho Chi Minh city (23/24 districts). The remaining participants were doctors and city-level governmental officers from Da Nang city, another important urban center of Vietnam.

The hypotheses were tested using structural equation modeling (SEM) by AMOS version 23 (IBM Inc.). This method is based on multiple regression equations which allow researchers to examine complex relationships among latent constructs (Schumacker and Lomax 2016). One of the advantages of SEM is that the outcome validation could be enhanced thank to the reliability of the constructs (Hoyle 1995). To do that, the author firstly conducted an exploratory factor analysis (EFA) to check the appropriation of the proposed scales. The process was followed by a confirmatory factor analysis (CFA). After all, the proposed hypotheses were tested using SEM. The full procedures for SEM applied in this study (summarized in Figure 2.2) were recommended by Gaskin (2016).

A full description of the materials and methods of study 1 is provided in Chapter 3.

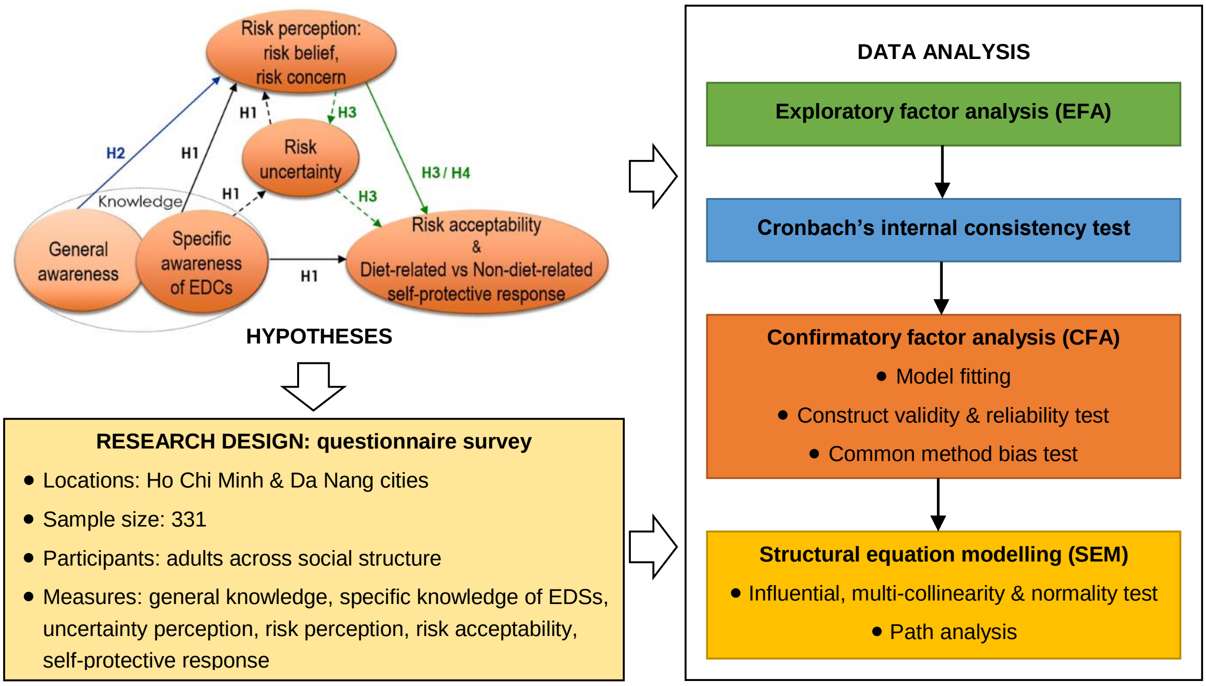

Figure 2.2 Methodological framework of study 1

2.2 Study 2: An integrated modelling framework for evaluating non-carcinogenic health risks from nonylphenol contaminated food consumption in Long An, Vietnam

The objective of the study was fulfilled though a five-stage process (described in Figure 2.3). First, the fate and distribution of NP in the Can Giuoc river is predicted using a fugacity-based multimedia model developed by Mackay (2001). Second, a food web bioaccumulation model (Arnot and Gobas 2004, Arnot et al. 2008) is adopted to estimate the equilibrium concentrations of NP, bioaccumulation factor and biomagnification factor for the species in the Can Giuoc river. Third, human exposure to NP is quantified using the accumulative concentrations in the biota and fishery products intake from a questionnaire survey among the local people. Next, a modified method for human health risk evaluation is proposed on the basis of the current method. Finally, the modified method is applied using data of intake frequency derived from the same survey.

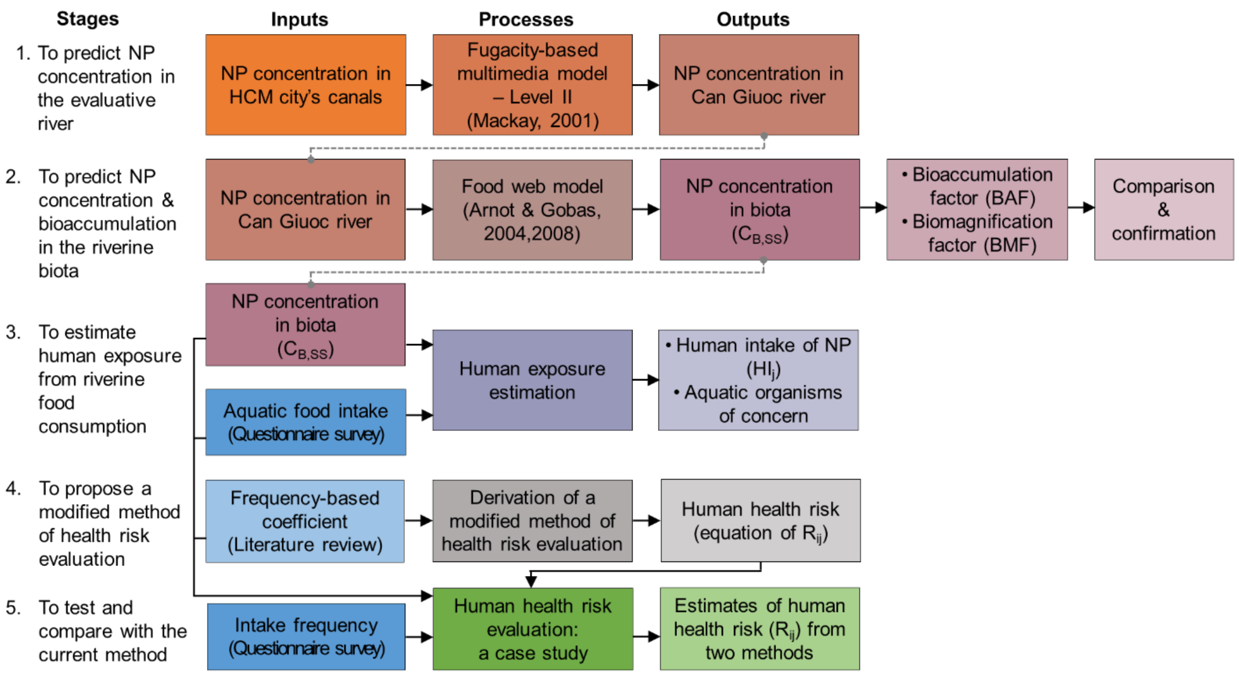

Figure 2.3 Methodological framework of study 2

The fugacity-based multimedia model (fate model)

This model is developed by Mackay (2001) based on the principles of fugacity at equilibrium and mass-balance. Mackay's models have been successfully and widely applied to predict the fate of persistent, bioaccumulative, and toxic substances (PBTs) under environmental risk assessment schemes. The evaluative environment is Can Giuoc river, which includes three compartments: water, suspended particulate matter (SPM), and sediment compartments, and to be considered as a steady-state system at equilibrium. An important source of inputs of this model was based on a two-year monitoring program on organic micro-pollutants in the aquatic environment in Vietnam conducted by Hanh (2015). The hydrological characteristics of the river were obtained from Mike 11, a modelling software package for rivers and channels developed by DHI Water & Environment, Denmark. The simulation of the river flow rate and velocity from 2010 to 2015 was conducted by the Southern Institute of Water Resources Research of Vietnam in July 2017 by the author's request under the scope of this study.

The food web bioaccumulation model (food web model)

The model is a combination of a food web bioaccumulation model (or BAF model) (Arnot and Gobas 2004) and a biotransformation model (Arnot et al. 2008). It was used for predicting the partitioning of NP into organisms, bioaccumulation and biomagnification factors for each species in the riverine food web. The models are able to realize the environmental and biological conditions, such as the availability of chemicals, water temperature, and kinetic mechanisms of chemical uptake and elimination in aquatic organisms (Gobas et al. 2009, Arnot and Gobas 2004, Arnot et al. 2008). Being verified with a large empirical data set, the models provide reasonable confidence and convenience for use (Arnot and Gobas 2004). Parameterizations of the food web and the model were literature-based, whilst the main inputs of this model were the outputs of the fate model.

Exposure and health risk evaluation

The method of evaluating human exposure and non-carcinogenic risk from NP contaminated riverine food consumption was adopted from USEPA (2000). The conventional processes were mainly based on the outputs of the food web model (e.g., concentrations of NP in biota) and data derived from a questionnaire survey (to be described in the next section). The modified method of NP risk evaluation was initiated from the findings of Huang et al. (2014), Chen et al. (2010), and particular Sise and Uguz (2017) that the level of NP in breast milk of mothers who consumed fish and used cleaning products at least once or twice per week was two-fold higher than that of those consuming fish and using cleaning products less frequently. This suggested the derivation of frequency-based coefficient indexes in accordance with given information of intake frequency. In order to test the modified method, intake frequency of riverine food derived from the questionnaire survey was integrated.

Questionnaire survey

The questionnaire survey was conducted in September 2017 among the adults dwelling in Can Giuoc and part of Can Duoc districts. Using semi open-ended questions, the survey aimed to collect the information on fishery products intake and intake frequency of eighteen riverine food products. The sample size was 203 people.

A full description of the materials and methods of study 2 is provided in Chapter 4.

2.3 Study 3: A firm behavior model based on a qualitative analysis of the Vietnamese textile firm's response to the restriction of nonylphenol and nonylphenol ethoxylates

Elaborating on extremely relevant theories of firm behavior, the main idea of this study is to propose a conceptual model for examining mechanisms of firm behavior. This was based on a qualitative analysis of technical specialists' perceptions of firm's organizational and technological adaptabilities, risks, benefits, and barriers, as well as their attitude, and an assessment of the current market situation and environmental regulations for the textile industry in Vietnam. The methodological framework of study 3 is presented in Figure 2.4.

Materials for the analysis were obtained from semi-structured in-depth interviews with five technical specialists: four from the textile firms (two small-medium-sized and two large-sized), and one from a transnational chemical supplier. Information on the market situation and environmental regulations for the textile industry in Vietnam was collected from a variety of data sources, including published and unpublished documents (e.g. scientific articles, project reports, regulatory documents) and electronic media (e.g. official websites, online news). Another source of information was via the un-structured talks with the interviewees, particularly with the expert from the chemical supplier. Cognitive mapping (Axelrod 2015) was employed to represent the relationships between the variables examined in this study.

More details of the materials and methods of study 3 is provided in Chapter 5.

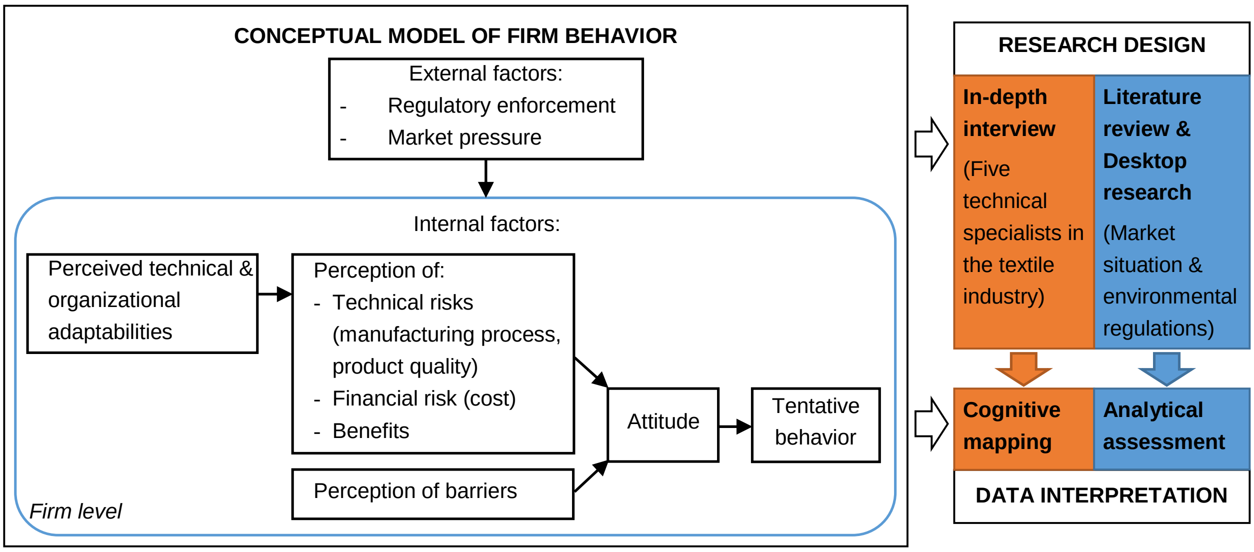

Figure 2.4 Methodological framework of study 3

2.4 Study 4: Distribution and removal of nonylphenol ethoxylates and nonylphenol from textile wastewater - A comparison of a cotton and a synthetic fiber factory in Vietnam

A cotton (F1) and a synthetic (F2) fiber factory in southern Vietnam were selected for the field investigation. They both include fabric/ fiber dyeing process and adopt a physico-chemical process followed by a biological process. The field investigation including a sampling campaign took place from September 5 to September 7, 2016. Operating parameters including design capacity, operating capacity, facility volume, hydraulic retention time (HRT), sludge returning schedule, and chemical use were provided by the operational staffs. The analysis of macro-compositions such as color, total suspended solids (TSS), mixed liquor suspended solids (MLSS), alkalinity, organic matter in terms of chemical oxygen demand (COD) and biological oxygen demand (BOD_5), ammonia ($N\text{-}NH_4^+$), nitrite ($N\text{-}NO_2^-$), and nitrate ($N\text{-}NO_3^-$) was conducted the laboratory of CENTEMA - Vietnam. NP and NPEOs were analyzed at the laboratory of the Foundation for Promotion of Material Science and Technology of Japan. The data was then subjected to the calculation of operating parameters such as solids retention time (SRT), returning activated sludge ratio, organic loading, and removal rate for the assessment of the distribution and removal efficiency. A methodological framework of study 4 is given in Figure 2.5.

A full description of the materials and methods of study 4 is provided in Chapter 6.

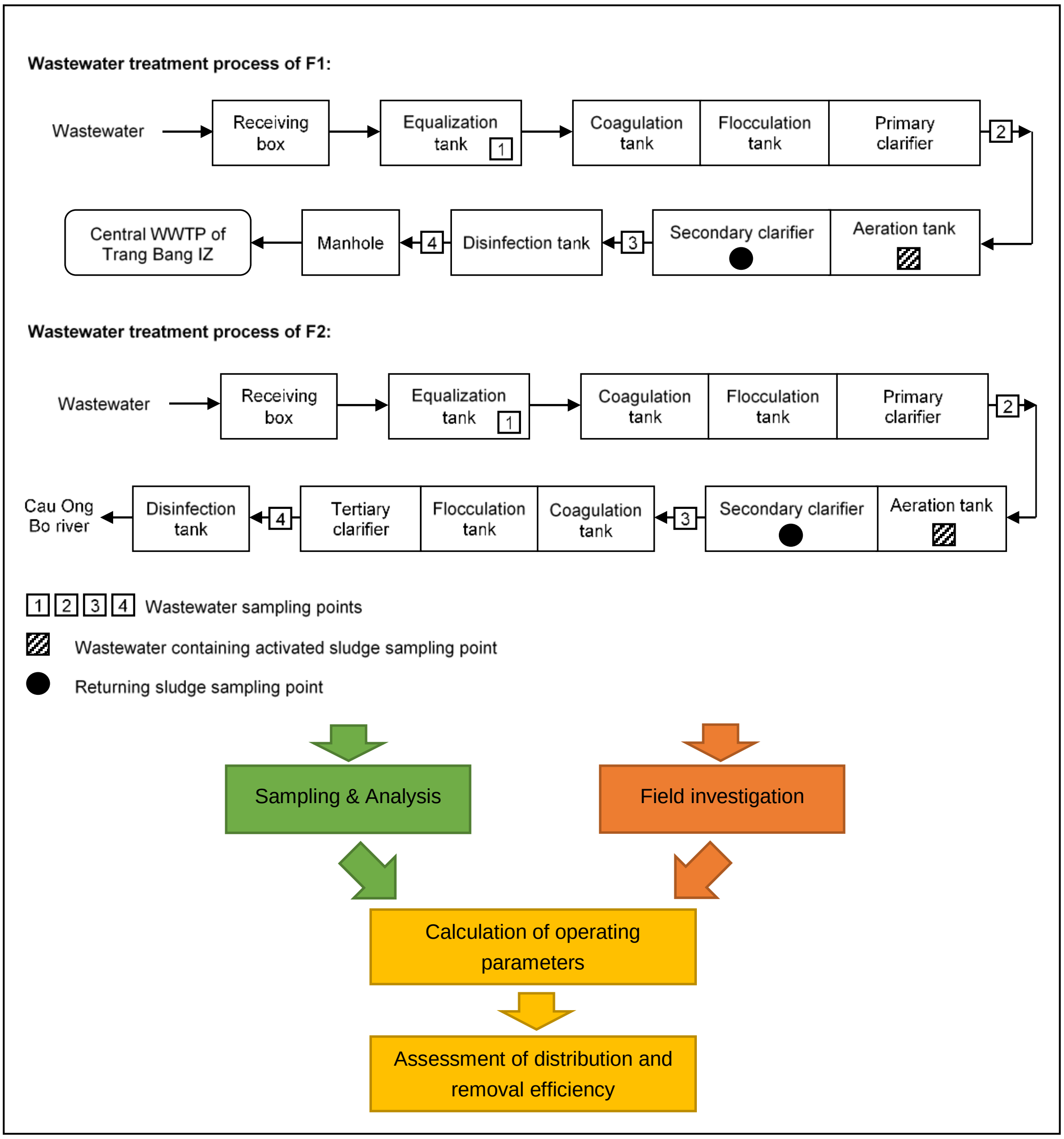

Figure 2.5 Methodological framework of study 4

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CHAPTER 3

SOCIETAL PERSPECTIVES AND SELF-PROTECTIVE BEHAVIOR TOWARDS EDSs

Summary

The ubiquitous surfactants nonylphenol (NP) and its ethoxylates (NPEOs), which are known as endocrine disrupters, have appeared in the lists of restricted chemical substances, monitoring programs, and environmental quality standards of many countries due to their adverse effects. Recent studies have reported alarming levels of NP, as the final metabolite of NPEOs, in Vietnamese urban waters, whilst response to this issue is negligible. With the aim of addressing how the public perceives and expects to avoid the risk of endocrine disrupting surfactants (EDSs), the study tested the hypothesized roles of specific knowledge, general knowledge, and perceived uncertainty using structural equation modelling. The findings revealed that different types of knowledge played certain roles in explaining risk perception, risk acceptability, and self-protective response, which are distinguished by experience among the public. Evidence of the mediating role that perceived uncertainty may play in the decrease of risk perception and the increase of risk unacceptance has been provided. The insights gained from the study may help answer why the public are in favor of taking non-diet-related self-protective measures rather than changing their dietary habits, which illustrates a comparison with the basis of health belief model. The needs for building cognitive capacity among the public, particularly pregnant women and young mothers, and risk communication concerning endocrine disrupting contamination linked to reproductive health are highlighted.

3.1 Introduction

The endocrine disrupting surfactants nonylphenol (NP) and its ethoxylates (NPEOs) are widely used in numerous industries such as cleansing, textile, personal care product, pesticide, paper, and plastic, and end up in the environment via municipal and industrial discharge (RPA and BRE 2003). As a final metabolite in the environment, NP has raised concern due to its endocrine disrupting effects on the reproductive, immune, and central nervous systems of wildlife and humans during the past two decades (Cosnefroy et al. 2009, Ghisari and Bonefeld-Jorgensen 2005, Jie et al. 2010, Kim et al. 2006, Mao et al. 2010, Razia et al. 2006, Soto et al. 1991, Vosges et al. 2012). Hormone-driven-characterized stages such as utero, infancy, childhood, puberty, and menopause are the most sensitive exposure periods (Crain et al. 2008, Gore et al. 2014, Muller 2013, Nepelska et al. 2014, Rachoń 2016). Regarding carcinogenesis, recent studies have shown the relation of exposure to NP to the risk of breast cancer in women (Wu et al. 2008) and prostate cancer in men (Forte et al. 2016, Kim et al. 2016).

Acknowledging the scientific uncertainty on the causal-effect concerning NP exposure in humans, regulations on chemical safety, restriction of marketing and use, environmental quality standard, and residue control have been promulgated with the aim of ecological and health protection within the European Union. Asian countries such as Taiwan, Korea, and Japan have also launched monitoring programs as well as proposed limiting NP in textile articles, especially for children. Nevertheless, although the concentration of NP in Vietnamese urban waters has reached alarming levels in recent years (Duong et al. 2010, Hanh et al. 2014, Minh et al. 2016, Viet et al. 2006), use of this chemical is still allowed and the response to ecological and health risk is negligible. This raises questions of how the society perceives the risk of these endocrine disrupting surfactants (risk perception), and how people expect to avoid the risk (risk acceptability and risk response). Answering these questions is practically vital for designing strategic response to the risk of endocrine disrupting compounds (EDCs) in Vietnam.

Objective: Taking into account the multiple applications of these endocrine disrupting surfactants (EDSs), this study focuses on their use in cleansing products, detergents, and cosmetics. The aim of the study is to examine the roles of the awareness of different levels and perceived uncertainty on EDSs risk perception, acceptability, and self-protective response among public using structural equation modeling (SEM). Hypotheses of this research are premised by the theories presented in the next section.

3.2 Materials and methods

3.2.1 Hypothesis formulation

In risk assessment process, understanding risk perception plays a key role in explaining risk acceptability and predicting risk response (Slovic 1987). How risk is perceived could be well explained by individual capacities such as knowledge (Agyei-Mensah and Oteng-Ababio 2012, Gaash et al. 2003, Kung and Lee 2006), or insight knowledge (Che et al. 2014), prior personal experience (Lindell and Hwang 2008), and self-controllability (McDaniels et al. 1997). Risk perception terminology captures one’s belief in negative consequences of an event as true (risk belief) and one’s concern about the consequences (risk concern) (Bish et al. 2011).

Theories of risk perception are initially developed on the basis of value-belief-norm (VBN) theory (Stern 2000, Stern et al. 1999) which intends to explain pro-environmental behavior. In expanded literature, the relationship of risk perception – risk acceptability predicting environmental action has been demonstrated by numerous scholars in the fields such as nuclear energy (De Groot and Steg 2010, De Groot et al. 2013, Peters and Slovic 1996), nuclear waste (Sjöberg and Drottz-Sjöberg 2001), flood risk due to climate change (De Dominicis et al. 2015), environmental hazard (Lindell and Hwang 2008), and ecological risk (Slimak and Dietz 2006). In this study, people’s self-protective response to health risk is of interest.

In order to make judgments on the risk of EDSs, people are expected to have specific knowledge about those chemicals such as their utilities, impacts, and pathways of exposure in addition to a general awareness of the surrounding environment and public health situation. As suggested by

Maxim et al. (2013), although people have rather good knowledge on a particular issue, they may feel uncertain or may have a perception of uncertainty (Brashers 2001, Powell et al. 2007) about that issue because they are not able to link the knowledge with the surrounding situation. However, the roles of specific knowledge compared with general awareness and perceived uncertainty regarding EDSs have not been well understood and proved by qualitative research. For these reasons, there is a need to qualitatively investigate three influences: public's cognition in terms of *general awareness* of water pollution and reproductive health problems (RHPs) and *specific awareness* of EDSs risk, as well as their *perceived uncertainty*.

The study is inspired from a publication on the health belief model (Rosenstock 1974) which provides a useful framework to explain public health behavior through four factors: perception of susceptibility, perception of severity, perception of barriers, and perception of benefits. It is suggested that perception of uncertainty is one of the barriers in decision-making (Darlow et al. 2016), thus it could be a barrier against behavioral changes. From that logic, it was hypothesized that perceived uncertainty might play a mediating role in the relationship of knowledge of EDSs with risk perception (in terms of risk belief and risk concern), as well as in the relationships of risk perception with risk acceptability and resulting self-protective response. Four hypotheses were formulated as follows (Figure 3.1):

Hypothesis 1 (H1): Perceived uncertainty mediates the positive relationships of specific awareness of EDSs with risk belief, risk concern, risk acceptability, and self-protective response.

Hypothesis 2 (H2): General awareness of water pollution and RHPs also has direct and positive effects on risk belief and risk concern.

Hypothesis 3 (H3): Perceived uncertainty plays a mediating role in the positive relationship of risk belief and risk concern with risk acceptability and self-protective response.

Hypothesis 4 (H4): People distinguished by experience and the status of pregnancy and child differ in the relationships of risk perception with risk acceptability and self-protective response.

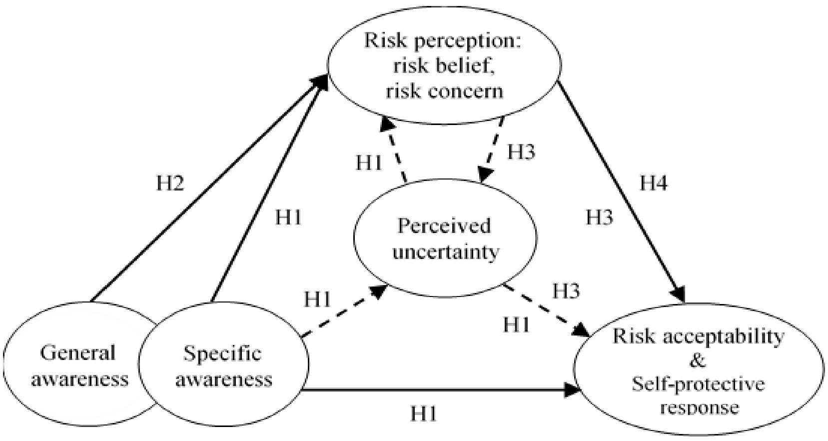

Figure 3.1 Hypothesized concept model

3.2.2 Research design

Data for this research was obtained randomly from 331 adult participants of all genders, including pregnant women and young mothers. The main study area is Ho Chi Minh city, which is the biggest urban center in Vietnam. Therefore, most of the participants who accounted for 92% of the sample were recruited across social structure in sub-urban and urban Ho Chi Minh city (23/24 districts). The remaining participants were doctors and city-level governmental officers from Da Nang city, another important urban center of Vietnam. Before the official data collection, close-ended questionnaires were pretested with 50 people to make sure all questions were clear and understandable. As a result, a definition of EDSs was added to help participants relate the questions about the chemicals to the information they have read or learnt. Participants were met at their homes in residential areas and worker houses, universities, research institutions, industrial and governmental agencies. Most of pregnant women and health care providers were accessed at hospitals. The questionnaires were self-administered.

3.2.3 Measures

Socio-demographic variables

Socio-demographic measures included age, gender, residence location, level of education, working field, marriage status, pregnancy and child status. An interval scale for age (18–25/ 26–40/ 41–60/ >60) and a nominal scale for gender (female/male) were used. The residence location of participants was recorded as one of 24 districts of Ho Chi Minh city or Da Nang city. Participants selected their education level from a nominal scale of junior high school/ high school/ undergraduate/ graduate/ postgraduate/ others. Working field included housewife/ worker/ student/ household business/ education or research (to be further specified by fields, such as biology, chemistry, environmental science)/medical and health care (e.g., doctor, health care technician, nurse)/ industrial manufacture/ service/ government (to be specified into sectors)/ non-governmental organization/ social organization/ other (self-indicated). Marriage status (single/ married), pregnancy status (pregnant/ non-pregnant/ tentatively pregnant), and child status (no child/ having at least one child: youngest child age of <6/ having at least one child: youngest child age of ≥6) were also recorded.

Experienced versus lay public

From the socio-demographic information, experienced public were defined based on their working fields. Accordingly, the experienced public were those working in the environmental governmental sector, research and education institutions (e.g., biological, chemical, environmental researchers, and lecturers), and medical and health care sector (e.g., doctors and health care technicians). It was assumed that they had information about the environmental situation, and biological-chemical-medical background that might be relevant to our research topic. Lay public were the remaining participants.

Pregnant women and young mothers versus the remaining respondents

Pregnant women and young mothers were distinguished using information on pregnancy and child status. They included women who were in pregnancy, those having at least one child at the age of less than 6 years old, and those planning to have a child.

General awareness of water pollution and reproductive health problems

This construct was derived from eight questions where the respondents selected their answers based on a five rating-point Likert scale from "totally disagree" to "totally agree". The first two questions asked about people's awareness of the intrinsic value of the nature, and whether it should be protected unconditionally. The second set of questions was to explore public's awareness of urban river and canal pollution, and whether that situation might adversely affect the health of human beings. Questions 5 and 6 aimed to investigate the publics' knowledge of some water pollutants that might cause RHPs such as pesticides, dioxins, bisphenol A, phthalates, heavy metals, and whether they knew about the increasing RHPs these days. The last two questions were relevant to pathways of exposure as food and drinking water produced from contaminated water sources that might cause RHPs.

Specific awareness of endocrine disrupting surfactants

The construct of specific awareness of EDSs was derived by six questions. The first question asked about public's cognition of the urban water contamination by EDSs. The next five questions explored public's knowledge in more details: EDSs and their sources, negative impacts as feminization in wildlife (fish) and reproductive disorders in humans, pathways of exposure via consuming riverine fish and drinking water. The generation of those questions was based on the knowledge of EDSs contamination (Duong et al. 2010, Hanh 2015, Hanh et al. 2014), the sources of EDSs (RPA and BRE 2003), and the effects on wildlife and humans (Å. Bergman et al. 2013). This construct adopted a mixed scale that measured the correctness of the participants' answers. The respondents were asked to make their judgments about the given information via seven options: "do not know", "wrong", "low certain", "moderately uncertain", "neutral (unsure)", "moderately certain", and "highly certain". It was supposed that people selected "do not know" because they had never heard or learnt about EDSs. Those who judged the statements as "wrong" might have known about EDSs but their knowledge might not be correct. Others who had certain knowledge of EDSs might select from "low certain" to "highly certain" depending on their confidence.

Risk perception

Public risk perception was examined through their belief (two questions) and concern about EDSs (two questions). The former questions asked about people's belief that EDSs in detergents, cleansing agents, and cosmetics might cause disorders of reproductive system in fish (e.g., feminization of fish) and disorders of reproductive system in humans. The latter investigated people's concern about the chemicals and the effects on humans, taking into account their selfcontrollabilities as not to consume riverine fish and to control from exposure to the chemicals. Respondents could select the best answers from a five rating-point Likert scale ("totally disagree" – "totally agree").

Perceived uncertainty

It was supposed that the public might feel uncertain about the impact of EDSs for some reasons. Therefore, in the corresponding questions, the uncertainty about the adverse effects on fish and humans was linked with three reasons: information about these surfactants has not appeared widely in media, insufficient attention and warning from scientists, and disorders in reproduction could be due to other reasons. This construct was measured by a five rating-point Likert scale ("totally disagree" – "totally agree").

Risk acceptability

Risk can be acceptable after it has been reduced further to be negligible (Kasperson 1983). Being aware that setting the level of acceptable risk was difficult and sensitive (Pasman 2015), indirect questions about what the government should do regarding mitigation or/and prevention of EDSs risk were adopted. The implied levels of controlling the concentration of EDSs from detergents, cleansing agents, and cosmetics in rivers were of interest. The first level was for the sake of aquatic life in general and riverine fish in particular (nature-oriented solution), whereas the second was for the human health and wellbeing (human-oriented solution). The third level aimed at a more prevention-oriented solution that required a control, giving that the adverse effects on human health were uncertain. A five rating-point Likert scale from "totally disagree" to "totally agree" was also adopted in this measurement.

Self-protective response

It was supposed that if people believed that they were exposed to EDSs via food and drink, they might take preventive measures to protect themselves. In this case, four options were suggested: to have frequent health checkup, to have a drinking water checkup, to install a drinking water filter, and to consume riverine fish less regularly. A similar five rating-point Likert scale as aforementioned was used for this measurement.

3.2.4. Analysis methods

The hypotheses were tested using structural equation modeling (SEM) by AMOS version 23 (IBM Inc., Chicago, IL, USA). This method was based on multiple regression equations which allowed researchers to examine complex relationships among latent constructs (Schumacker and Lomax 2016). One of the advantages of SEM was that the outcome validation could be enhanced thank to the reliability of the constructs (Hoyle 1995). To do that, an exploratory factor analysis (EFA) was conducted to check the appropriation of the proposed scales. The process was followed by a confirmatory factor analysis (CFA). CFA served as an intermediary but essential step to provide supports to the SEM validation. After all, the proposed hypotheses were tested using SEM. The full procedures for SEM applied in this study were recommended by Gaskin (2016).

Exploratory factor analysis procedures

The sample size was 331 and missing values of each variable of maximum 2% were replaced by median values for ordinal variables. An EFA was conducted for 27 items using maximum likelihood extraction method and subjected to a varimax rotation. This factor analysis suggested a seven-factor solution with the accumulative extraction sum of squared loadings of 61.2%. Kaiser-Meyer-Olkin (KMO) value of the test was 0.84, which indicated that the observed correlation matrices were factorable. The KMO measure of sampling adequacy of greater than 0.5 is acceptable to proceed a factor analysis (Kaiser 1974). Based on the loadings of items, the following modifications were made:

- The scale "general awareness of water pollution and RHPs" was split into two, which were named as "general awareness of water pollution and RHPs" and "awareness of the pathways of exposure that might affect reproductive health", comprising of six and two items, respectively (Figure 3.2). Since the items of the latter scale may require more insight knowledge compared with those in the former scale, this modification is acceptable.
- "Risk belief" items of the "risk perception" scale showed an unclear trend of loading on two factors. Therefore, "risk belief" items were treated as components of a separate factor in the following CFA.
- The first three items of the "self-protective response" scale showed rather good loadings on one factor. Since these questions were irrelevant to food, the factor was named as "non-diet-related self-protective response". The last item of this scale particularly relevant to a change in daily riverine fish diet presented a loading on none of the factors, so it was treated as a "stand-alone" endogenous variable in CFA, namely as "diet-related self-protective response".

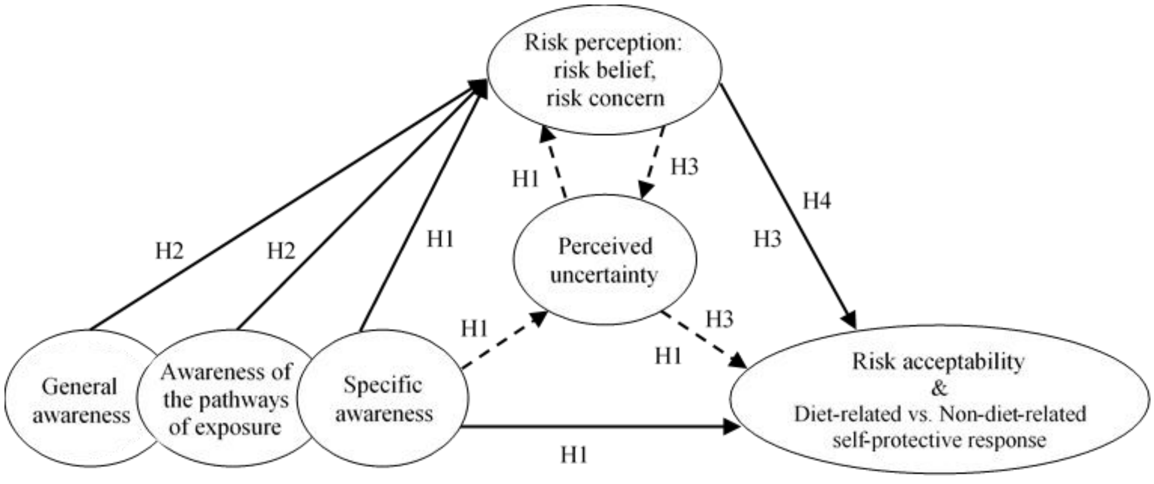

Figure 3.2 Modified hypothesized concept model

Items of the remaining scales showed good loadings. All factor loadings (standardized regression weights) were statistically significant ($p < 0.001$). Next, based on the EFA results and substantive interpretation, internal consistency of each scale was examined using Cronbach's alpha. All scales showed high Cronbach's alphas of from 0.79 to 0.91, which indicated good internal consistency. The alpha coefficients of 0.70 and above indicate sufficient reliability "in the early stages of research on predictor tests or hypothesized measures of a construct" (Nunnally 1978). The results of EFA were summarized in Table 3.1.

It was suggested from the EFA that awareness was composed of three levels (three factors), risk perception including risk belief and risk concern (two factors), and self-protective response including "non-diet-related self-protective response" (a factor) and "diet-related self-protective response" (a "stand-alone" endogenous variable). The modification of the hypothesized model was illustrated in Figure 3.2.

Table 3.1 Alpha coefficients and factor loadings (n = 331)

<table>
<tr><th colspan="2">Construct</th><th colspan="2">Variable</th><th>Standardized regression weight</th></tr>
<tr><td colspan="2" rowspan="6">General awareness of water pollution and reproductive health problems (GA) (α = 0.87)</td><td>GA1</td><td>I believe that the nature is valuable for its own sake.</td><td>0.643</td></tr>
<tr><td>GA2</td><td>I believe that to protect the environment unconditionally is important.</td><td>0.829</td></tr>
<tr><td>GA3</td><td>The rivers and canals in our cities are polluted at different levels.</td><td>0.728</td></tr>
<tr><td>GA4</td><td>Polluted rivers may have adverse effects on the health of human beings.</td><td>0.851</td></tr>
<tr><td>GA5</td><td>Some water pollutants that may negatively impact the reproductive health of human beings are pesticides, dioxins, bis phenol A, phthalate, heavy metals (e.g., Cd, Pb, Hg)</td><td>0.640</td></tr>
<tr><td>GA6</td><td>More people have reproductive health problems these days.</td><td>0.726</td></tr>
<tr><td colspan="2" rowspan="2">Awareness of the pathways of exposure that may affect reproductive health (AP) (α = 0.88)</td><td>AP1</td><td>Reproductive health problems are possibly caused by food produced from polluted water.</td><td>0.931</td></tr>
<tr><td>AP2</td><td>Reproductive health problems are possibly caused by drinking water exploited from contaminated water resources.</td><td>0.841</td></tr>
<tr><td colspan="2" rowspan="6">Specific awareness of endocrine disrupting surfactants (SA) (α = 0.85)</td><td>SA1</td><td>These days, urban waters are contaminated by EDCs discharged from industrial, agricultural, and domestic activities.</td><td>0.618</td></tr>
<tr><td>SA2</td><td>Consuming detergents, cleansing agents, or cosmetics, some industries such as textile, paper, cleansing and domestic activities discharge wastewater that contains EDSs.</td><td>0.714</td></tr>
<tr><td>SA3</td><td>EDSs in detergents, cleansing agents, cosmetics, etc. may cause feminization in wildlife (fish).</td><td>0.661</td></tr>
<tr><td>SA4</td><td>One of the reasons of reproductive disorders in human (e.g., abnormalities in fetus development, testicular dysgenesis syndrome (TDS) in men), is being exposed to EDSs in commonly domestic products such as detergents, cleansing agents, cosmetics, etc.</td><td>0.779</td></tr>
<tr><td>SA5</td><td>People may uptake these chemicals via consuming bio-accumulated riverine fish.</td><td>0.668</td></tr>
<tr><td>SA6</td><td>People may uptake these chemicals via drinking water exploited from contaminated water resources although the water is treated.</td><td>0.676</td></tr>
<tr><td rowspan="4">Risk perception</td><td rowspan="2">Risk belief (RB) (α = 0.80)</td><td>RB1</td><td>I believe that EDSs in commonly domestic products such as detergents, cleansing agents, cosmetics may cause reproductive abnormalities in fish (e.g., feminization of fish).</td><td>0.803</td></tr>
<tr><td>RB2</td><td>I believe that EDSs in commonly domestic products such as detergents, cleansing agents, cosmetics may cause reproductive abnormalities in humans.</td><td>0.825</td></tr>
<tr><td rowspan="2">Risk concern (RC) (α = 0.84)</td><td>RC1</td><td>I concern about those chemicals and their effects on humans although I do not consume riverine fish.</td><td>0.805</td></tr>
<tr><td>RC2</td><td>I concern about those chemicals and their effects on humans although I can control my exposure to them.</td><td>0.893</td></tr>
<tr><td colspan="2" rowspan="2">Perceived uncertainty (UN) (α = 0.91)</td><td>UN1</td><td>I am uncertain about the adverse effects on fish and humans because information about EDSs appears in just a few sources.</td><td>0.898</td></tr>
<tr><td>UN2</td><td>I am uncertain about the adverse effects on fish and humans because of insufficient attention and warning from scientists.</td><td>0.930</td></tr>
</table>

Construct		Variable	Standardized regression weight
	UN3	I am uncertain about the adverse effects on fish and humans because disorders in reproduction can be relevant to other reasons.	0.801
Risk acceptability (RAC) (α = 0.89)	RAC1	I suggest that the level of EDSs from detergents, cleansing agents, cosmetics, etc. in rivers should be controlled for their adverse effects on aquatic life in general, and on riverine fish in particular.	0.912
	RAC2	I suggest that the level of EDSs from detergents, cleansing agents, cosmetics, etc. in rivers should be controlled for their possible effects on humans.	0.858
	RAC3	I suggest that the level of EDSs from detergents, cleansing agents, cosmetics, etc. in rivers should be controlled even though the adverse effects on the health of humans are uncertain.	0.798
Non-diet-related self-protective response (NDSP) (α = 0.79)	SP1	I am thinking of having a frequent health checkup.	0.696
	SP2	I am thinking of having a drinking water checkup.	0.831
	SP3	I am thinking of installing a drinking water filter for my family.	0.722
Diet-related self-protective response [i] (DSP)	SP4	I suppose to consume riverine fish less regularly.	

Note: [i] "Stand-alone" endogenous variable.

Confirmatory factor analysis procedures

The CFA procedures are proposed by Gaskin (2016). This process included four main steps: (1) model fit; (2) examining the validity and reliability of the factors in CFA test; (3) checking common method bias, and (4) testing measurement model invariance across groups. Maximum likelihood estimation method was used because it could work with light to moderate skewness and kurtosis with rather small sample size (100 – 400 subjects) (Reisinger and Mavondo 2007). Goodness of model fit was assessed based on Chi-square/*df* ratio (CMIN/*df*), Comparative Fit Index (CFI), Tucker Lewis Index (TLI), Standardized Root Mean Square Residual (SRMR), Root Mean Square Error of Approximation (RMSEA), and *p* of Close Fit (PCLOSE). The CFI and TLI values of greater than 0.90 reflect an acceptable fit to good fit (≥ 0.95) (Bentler and Bonett 1980, Hu and Bentler 1999). RMSEA values of from 0.05 to 0.08 indicate an acceptable fit and those of below 0.05 represent a good fit, whereas SRMR values of less than 0.08 generally show a good fit (Hu and Bentler 1999). The PCLOSE tests the null hypothesis that the RMSEA is of 0.05, therefore PCLOSE values of greater than 0.05 indicate close-fitting models (Byrne 2013). The results of CFA were summarized in Table 3.2 and Appendix 1: Tables A1.1 – A1.2.

Table 3.2 Model fit indices ([i] n = 331; [ii] n = 328)

Model	χ^2	*df*	CMIN/*df*	CFI	TLI	SRMR	RMSEA	PCLOSE
Initial measurement model [i]	675.0	293	2.3	0.92	0.91	0.06	0.06	<0.010
CLF unconstrained model [i]	527.1	266	2.0	0.95	0.93	0.05	0.05	0.135
χ^2 and df difference	$\Delta\chi^2$ = 147.9	Δdf = 27						
Measurement model controlled by experience								
Unconstrained model [i]	1116.7	605	1.8	0.90	0.89	0.07	0.05	0.396
Measurement weights constrained model [i]	1097.4	586	1.9	0.90	0.88	0.07	0.05	0.294
χ^2 and df difference	$\Delta\chi^2$ = 19.3	Δdf = 19						
Measurement model controlled by pregnancy and child status								
Unconstrained model [i]	1113.6	605	1.8	0.90	0.89	0.07	0.05	0.418
Measurement weights constrained model [i]	1076.3	586	1.8	0.91	0.89	0.07	0.05	0.434
χ^2 and df difference	*$\Delta\chi^2$ = 37.3*	Δdf = 19						
Structural weights model [ii]	11.6	8	1.4	0.99	0.98	0.03	0.04	0.636
Structural weights model controlled by experience								
Unconstrained model [ii]	22.1	16	1.4	0.99	0.97	0.04	0.03	0.764
Structural weights constrained model [ii]	90.2	42	2.1	0.94	0.89	0.11	0.06	0.172
χ^2 and df difference	$\Delta\chi^2$ = 68.1	Δdf = 26						

The initial measurement model achieved good model fit with χ^2 = 675.0, df = 293; CMIN/df = 2.3, CFI = 0.92, TLI = 0.91, SRMR = 0.06, and RMSEA = 0.06, whereas the PCLOSE test was statistical significant (p < 0.01), which suggested that the initial model was not so close-fitting (Table 3.2). Modification indices that suggested covariances in three pairs of observed variables were accepted: GA5 and GA6 were both about general awareness of RHPs, SA1 and SA2 both relevant to knowledge of EDSs sources, and SA5 and SA6 investigating knowledge of pathways of exposure to EDSs.

Construct reliability and validity were assessed using criteria suggested by Hair (2010). The latent constructs achieved a composite reliability (CR) since all CR values exceeded 0.6 (CV = 0.8 – 0.9). Most of the average variance extracted (AVE) values were greater than 0.5 except for the construct of specific awareness of EDSs (AVE = 0.5), indicating an acceptable reliability of the measurement model in measuring the constructs. The maximum shared variance (MSV) values and the average shared variance (ASV) values of the constructs were smaller than the AVE values, and the square root of the AVE values were greater than the inter-construct correlation coefficients. Therefore, the discriminant validity for all constructs was also achieved. Details of the construct reliability and validity, and the construct discriminant validity were provided in Appendix 1: Tables A1.1 and A1.2, respectively.

The next step aimed to check common method bias using "controlling for the effects of a single unmeasured latent method factor" (Podsakoff et al. 2003). A first-order common latent factor (CLF) was added into the measurement model. A chi-square test for the difference between the CLF unconstrained model and the CLF full-constrained model was conducted. The resulting chi-square difference was statistically significant ($\Delta\chi^2 = 147.9$, $\Delta df = 27$, $p < 0.01$). Since the CLF had significant shared variance, it was retained in the baseline model. The unconstrained model where the variance of CLF was set to 1 had improved fitting indices: $\chi^2 = 527.1$, $df = 266$; CMIN/df of 2.0, CFI = 0.95, TLI = 0.93, SRMR = 0.05, RMSEA = 0.05, and PCLOSE = 0.135 (Table 3.2). Factor scores that accounted for the shared variances of the CLF were imputed from CLF unconstrained model.

Configural invariance tests for the measurement model controlled by experience and status of pregnancy and child obtained an adequate goodness of fit (Table 3.2). Metric invariance test for the measurement model controlled by pregnancy and child status revealed that forcing two groups together was substantially different than letting them be estimated freely ($\Delta\chi^2 = 37.3$, $\Delta df = 19$, $p < 0.01$). Whereas, the metric invariance test for the measurement model controlled by experience resulted in a non-significant chi-square difference ($\Delta\chi^2 = 19.3$, $\Delta df = 19$, $p = 0.438$), indicating that the groups were non-significantly different at measurement model level; however, they could be different at path level. Therefore, it was suggested to check structural weights model invariance and path differences between groups.

Structural equation modelling procedures

Before conducting SEM, multivariate assumptions including influential, multicollinearity, and normality by groups were examined (see Appendix 1: Tables A1.3 – A1.4). Three records (Cook's distance of from 0.13 to 0.21) that deviated much far from others were removed in order to eliminate their influences on the results. The new sample size was 328 (331 − 3 = 328). The multicollinearity test resulted in the tolerance values of 0.60 – 0.86 and the VIF values of 1.2 – 1.7, which were satisfactory (tolerance > 0.1 and VIF < 3). A Shapiro-Wilk normality test ($p < 0.01$) was performed on eight factors and one dependent variable showed a statistically significant non-normality of the data set. A visual inspection of their histograms, normal Q-Q plots and box plots across groups revealed light to moderate deviations from normal distributions. The z-values of the skewness for the experienced public, the lay public, the pregnant women and young mothers, and the remaining population were in the ranges of -5.6 – 1.6, -8.8 – 3.8, -8.2 – 3.2, and -8.8 – 2.4, respectively, and the z-values of the kurtosis were -1.8 – 3.7, -3.4 – 5.6, -2.7 – 8.2, and -3.3 – 5.2, respectively. As an exception, "general awareness of water pollution and RHPs" construct had extreme z-values of skewness from -8.8 to -15.0 and z-values of kurtosis from 11.0 to 27.8.

Imputed factor scores from CLF unconstrained measurement model and maximum likelihood method of estimation were used in our path model. After accepting the covariances between GA and AP, and between e3 and e4 (the residuals of "risk acceptability" and "non-diet-related self-protective response", respectively, the baseline path model had significantly improved in goodness of fit with χ^2 = 11.6, df = 8; p = 0.172, CMIN/df = 1.4, CFI = 0.99, TLI = 0.98, SRMR = 0.03, RMSEA = 0.04, and PCLOSE = 0.636 (see Table 3.2). The shared variances were understandable because the former pair of constructs might be relevant to a more global concept of environmental pollution and public health, whereas the latter pair of constructs might be affected by a certain factor. Afterwards, the hypothesized relationships (H1, H2, and H3) were tested. The level of confidence of the hypothesis tests was set at 95% (α = 0.05). Testing of hypothesis H4 was conducted in two stages. First, group comparisons (multi-sample approach) suggested by Reinecke (1999) were made in order to obtain an insight of the structural weights invariances using the same baseline model. The invariance test revealed a significant difference between the experienced and the lay public at structural weights model level ($\Delta\chi^2$ = 68.1, Δdf = 26; p < 0.01), whereas the model controlled by pregnancy and child status showed a non-significant difference between the groups ($\Delta\chi^2$ = 25.3, Δdf = 26; p = 0.501). Since those invariance tests were still at model level, parameter estimates invariances in single relationships of risk perception with risk acceptability and with self-protective response between groups were examined.

3.3 Results

3.3.1 Participants

The sample was representative by early adults (26 – 40: 54.4%), young adults (18 – 25: 37.9%), middle adults (41 – 60: 6.7%), and latter adults (>60: 0.9%). Female and male rates were 64.4% and 35.6%, respectively. Most of the participants achieved undergraduate level of education, a rate of 73.5%, whereas the shares of junior high school, high school, graduate, postgraduate, and others were 2.8%, 12.3%, 7.7%, 0.6%, and 3.1%, respectively. Among the participants living in Ho Chi Minh city, who accounted for 92.2% of the sample, 67.4% were from urban and 24.8% were from suburban districts. Only 7.8% of the participants were from Da Nang city. Based on the aforementioned criteria of experience, 29% of the participants were categorized as experienced public, and 71% were defined as lay public. The share of pregnant women and young mothers was 36%, and the remaining group accounted for 64% of the sample. The numbers of missing value were one for age, two for gender, four for education, and six for residence location in the total of 328 records.

3.3.2 Mean values and bivariate correlations

The minimum, maximum, and mean values of each construct were presented in Tables 3.3. The results in Table 3.3 revealed a high levels of general awareness of water pollution and RHPs (GA: μ = 5.2 – 5.4), risk acceptability (RAC: μ = 5.3 – 5.6), and non-diet-related self-protective response (NDSP: μ = 5.2 – 5.5) among the public. The levels of awareness of the pathways of exposure that may affect reproductive health (AP: μ = 3.7 – 3.8), risk belief (RB: μ = 3.8 – 4.1), and risk concern (RC: μ = 4.0 – 4.1) were moderately high. Whereas, the public had rather low level of specific awareness of EDSs (SA: μ = 1.7 – 1.8) with neutral uncertainty (UN: μ = 1.9 – 2.1), and they showed neutral tendency of diet-related self-protective response (DSP: μ = 2.7 – 3.0).

A t-test analysis for experience revealed that GA, RB, RAC, NDSP for the lay public were statistically significantly higher, but DSP was lower than those for the experienced public. A t-test analysis for the status of pregnancy and child showed the only significant difference in RU. The *df* values, *t*-values, and *p*-values were provided in Table 3.4.

Because SEM was based on Pearson correlation (Schumacker and Lomax 2016) whereas the data set showed deviations from normality across constructs, both Pearson correlation (r) and Spearman's rho (r_s) were examined. The parametric and non-parametric estimations resulted in comparative bivariate correlations in terms of magnitude and statistical significance (see Table 3.5). The results in Table 3.5 revealed strong correlations of RAC with GA ($r = 0.427$, $r_s = 0.454$, $p < 0.01$) and RB ($r = 0.478$, $r_s = 0.362$, $p < 0.01$), and a weaker relationship with RC ($r = 0.338$, $r_s = 0.330$, $p < 0.01$). Meanwhile, the correlations of RAC with AP and SA were statistically non-significant. NDSP was positively and significantly related to three levels of awareness and risk perception where the relationship with RB was strongest ($r = 0.472$, $r_s = 0.486$, $p < 0.01$). However, DSP showed non-significant relationships with the constructs of awareness and risk perception. Whilst UN was negatively associated with NDSP ($r = -0.210$, $r_s = -0.252$, $p < 0.01$), it was positively correlated with DSP ($r = 0.200$, $r_s = 0.211$, $p < 0.01$).

Table 3.3 Mean values (n = 331)

Construct		Group	Min. – Max. [i]	Mean [i] – μ (σ)
General awareness of water pollution and reproductive health problems (GA)		Inexperienced	1.4–6.5	5.4 (0.05)
		Experienced	1.2–6.4	5.2 (0.10)
		N_Pr.Ym	1.2–6.5	5.4 (0.06)
		Pr.Ym	1.4–6.4	5.4 (0.07)
Awareness of the pathways of exposure that may affect reproductive health (AP)		Inexperienced	0.5–5.6	3.8 (0.06)
		Experienced	0.9–5.5	3.7 (0.09)
		N_Pr.Ym	0.5–5.6	3.7 (0.06)
		Pr.Ym	0.9–5.4	3.8 (0.08)
Specific awareness of endocrine disrupting surfactants (SA)		Inexperienced	0.1–3.5	1.8 (0.06)
		Experienced	0.1–3.5	1.7 (0.09)
		N_Pr.Ym	0.1–3.5	1.8 (0.06)
		Pr.Ym	0.3–3.5	1.8 (0.08)
Risk perception	Risk belief (RB)	Inexperienced	1.7–6.0	4.1 (0.05)
		Experienced	1.3–5.6	3.8 (0.10)
		N_Pr.Ym	1.3–6.0	4.1 (0.06)
		Pr.Ym	1.7–5.8	4.0 (0.08)
	Risk concern (RC)	Inexperienced	0.7–4.9	4.0 (0.06)
		Experienced	0.8–4.9	4.0 (0.09)
		N_Pr.Ym	0.7–4.9	4.0 (0.07)
		Pr.Ym	2.0–4.9	4.1 (0.07)
Perceived uncertainty (UN)		Inexperienced	0.7–4.1	2.0 (0.06)
		Experienced	0.6–4.1	2.1 (0.09)
		N_Pr.Ym	0.7–4.1	2.1 (0.07)
		Pr.Ym	0.6–4.0	1.9 (0.08)
Risk acceptability (RAC)		Inexperienced	2.6–6.9	5.6 (0.05)
		Experienced	1.5–6.8	5.3 (0.12)
		N_Pr.Ym	2.6–6.9	5.5 (0.06)
		Pr.Ym	1.5–6.9	5.5 (0.09)
Non-diet-related self-protective response (NDSP)		Inexperienced	2.5–6.5	5.5 (0.05)
		Experienced	1.8–6.5	5.2 (0.11)
		N_Pr.Ym	2.3–6.5	5.4 (0.06)
		Pr.Ym	1.8–6.5	5.5 (0.08)
Diet-related self-protective response (DSP)		Inexperienced	1.0–5.0	2.7 (0.07)
		Experienced	1.0–5.0	3.0 (0.11)
		N_Pr.Ym	1.0–5.0	2.8 (0.08)
		Pr.Ym	1.0–5.0	2.7 (0.11)

Notes: [i] Min., max., and mean values that accounted for the shared variances of the CLF were imputed from CLF unconstrained model. σ indicates standard deviation. Inexperienced: lay public; Experienced: experienced public; Pr.Ym: pregnant women and young mothers; N_Pr.Ym: the remaining population.

Table 3.4 Difference in mean values (n = 331)

Construct		t-test for experience		t-test for the status of pregnancy and child	
		t value	*p* value	*t* value	*p* value
General awareness of water pollution and reproductive health problems (GA)		4.322	0.038	0.006	0.938
Awareness of the pathways of exposure that may affect reproductive health (AP)		0.257	0.612	0.534	0.465
Specific awareness of endocrine disrupting surfactants (SA)		0.791	0.374	0.181	0.671
Risk perception	Risk belief (RB)	7.280	0.007	1.462	0.228
	Risk concern (RC)	0.011	0.916	1.284	0.258
Perceived uncertainty (UN)		0.846	0.358	4.891	0.028
Risk acceptability (RAC)		7.185	0.008	0.005	0.942
Non-diet-related self-protective response (NDSP)		7.308	0.007	0.810	0.369
Diet-related self-protective response (DSP)		5.172	0.024	0.388	0.534

Table 3.5 Bivariate correlations (n = 331)

Construct		Pearson correlation – r (Spearman's rho – r_s)							
		GA	AP	SA	RB	RC	UN	RAC	NDSP
General awareness of water pollution and reproductive health problems (GA)									
Awareness of the pathways of exposure that may affect reproductive health (AP)		0.331 ** (0.133 *)							
Specific awareness of endocrine disrupting surfactants (SA)		−0.078 (−0.073)	0.014 (0.053)						
Risk perception	Risk belief (RB)	0.249 ** (0.168 **)	0.225 ** (0.223 **)	0.529 ** (0.548 **)					
	Risk concern (RC)	0.337 ** (0.428 **)	0.057 (0.023)	0.184 ** (0.232 **)	0.134 * (0.171 **)				
Perceived uncertainty (UN)		−0.009 (0.047)	−0.084 (−0.118 *)	−0.400 ** (−0.393 **)	−0.317 ** (−0.357 **)	−0.403 ** (−0.498 **)			
Risk acceptability (RAC)		0.427 ** (0.454 **)	0.029 (−0.045)	0.084 (0.056)	0.478 ** (0.362 **)	0.338 ** (0.330 **)	0.066 (0.114 *)		
Non-diet-related self-protective response (NDSP)		0.204 ** (0.201 **)	0.191 ** (0.226 **)	0.357 ** (0.362 **)	0.472 ** (0.486 **)	0.137 * (0.157 **)	−0.210 ** (−0.252 **)	0.520 ** (0.474 **)	
Diet-related self-protective response (DSP)		−0.046 (−0.031)	−0.052 (−0.043)	−0.057 (−0.062)	−0.083 (−0.096)	−0.128 * (−0.131 *)	0.200 ** (0.211 **)	−0.002 (0.020)	−0.068 (−0.119 *)

Note: * Correlation is significant at the 0.05 level (2-tailed); ** Correlation is significant at the 0.01 level (2-tailed).

3.3.3 Hypothesis 1

The outcome model and the results from the path analysis were summarized in Figure 3.3 and Table 3.6, respectively. The model was composed of eight constructs and one observed endogenous variable. Only statistically significant standardized regression weights (β) were shown, whereas paths without beta indicated non-significant effects.

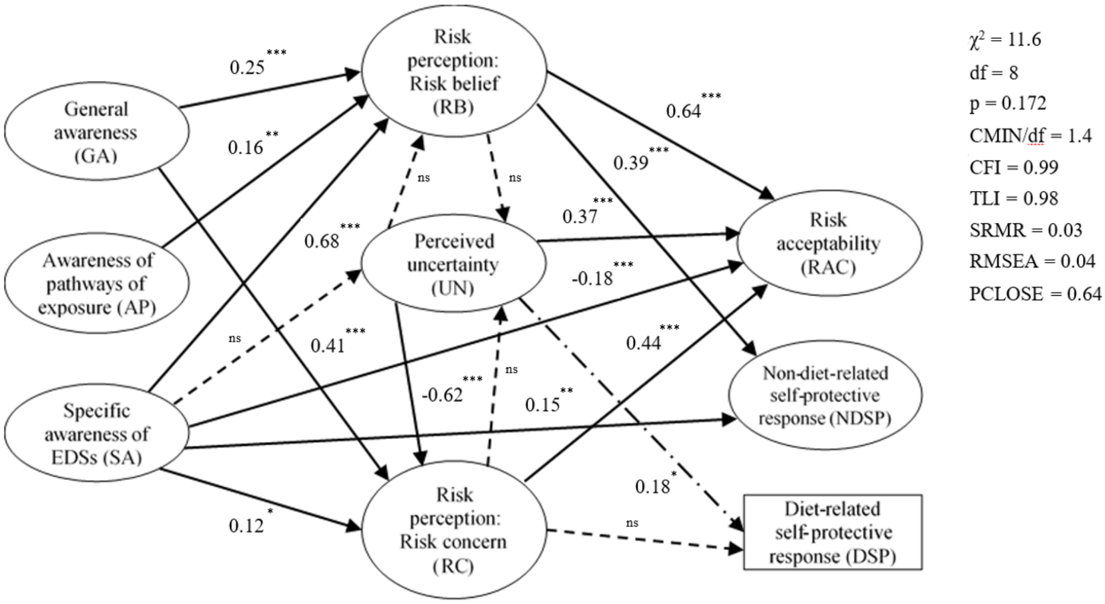

Figure 3.3 Final model of the determinants of risk perception, risk acceptability, and self-protective response

Notes: Direct effects: thick solid arrows; moderating effects: slim solid and dot lines; indirect effects: combinations of a slim dot line (SA/RB to UN) and a long dash dot line (UN to DSP).

* $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$; ^{ns} non-significant.

Table 3.6 Summary of the hypotheses and outcomes (n = 328)

Hypothesis	Outcome	Conclusion
H1: Perceived uncertainty (UN) mediates the positive relationships of specific awareness of endocrine disrupting surfactants (SA) with risk belief (RB), risk concern (RC), risk acceptability (RAC), and self-protective response (NDSP & DSP).	SA → RB, ↘UN↗ Partial mediation SA → RC, ↘UN↗ Partial mediation SA → RAC, ↘UN↗ Partial mediation	Supported H1 for risk belief and risk concern; partially supported H1 for risk acceptability
	SA → NDSP Direct effect SA DSP, ↘UN↗ Indirect effect	Did not support H1 for self-protective responses
H2: General awareness of water pollution and reproductive health problems (GA & AP) also has direct and positive effects on risk belief (RB) and risk concern (RC).	GA → RB Direct effect GA → RC Direct effect AP → RB Direct effect	Supported H2
	AP —— RC Non-significant effect	Did not support H2
H3: Perceived uncertainty (UN) plays a mediating role in the positive relationship of risk belief (RB) and risk concern (RC) with risk acceptability (RAC) and self-protective response (NDSP & DSP).	RB → RAC, ↘UN↗ Partial mediation RC → RAC, ↘UN↗ Partial mediation RC → DSP, ↘UN↗ Full mediation	Supported H3
	RB → NDSP Direct effect RB DSP, ↘UN↗ Indirect effect RC —— NDSP Non-significant effect	Did not support H3
H4: People distinguished by "experience" and "status of pregnancy and child" differ in the relationships of risk perception (RB) with risk acceptability (RAC) and self-protective response (NDSP & DSP).	Difference by "experience": RB → RAC ($\Delta\chi^2$ = 5.0, Δdf = 1; p = 0.025) RB → NDSP ($\Delta\chi^2$ = 4.6, Δdf = 1; p = 0.031)	Supported H4
	Non-significant difference by "status of pregnancy and child"	Did not support H4

The findings revealed that the effects of specific awareness of EDSs (SA) on risk belief (RB) and risk concern (RC) completely supported H1. Accordingly, the paths from SA to RB (β = 0.68, z = 4.83), and to RC (β = 0.12, z = 1.62) were positive and statistically significant. SA appeared to have a strong association with RB, suggesting that the insight of EDSs was a determinant of perceiving EDSs risk. In addition, this relationship was weakened in the presence of an uncertain feeling about EDSs risk, indicating the mediating role of perceived uncertainty (UN). Indeed, when the relationship was separately investigated, the direct effects without- and with-mediation were 0.54 (p < 0.01) and 0.48 (p < 0.01), respectively. Indirect effects of SA on RB through UN were non-significant, thus they were not shown in Figure 3.3.

Nonetheless, these results seemed not similar for RC. SA slightly associated with RC with a partial mediation through UN. Among the indirect relationships, only the path from UN to RC was significant ($\beta = -0.62$, $z = -3.33$). The negative beta implied that the less uncertainty about EDSs risk, the more concerning about consequences people might feel.

The direct association of SA with risk acceptability (RAC) was marginally negative and significant ($\beta = -0.18$, $z = -3.62$). It was partially mediated by UN, which exerted a positive and significant influence on RAC ($\beta = 0.37$, $z = 7.70$). These inverse trends could be understood in a way that people who were more ambiguous about EDSs risk showed more in favor of supporting EDSs controlling strategies or the unlikeliness of risk acceptance. This partially supported H1.

Whist SA successfully explained non-diet-related self-protective response (NDSP) with a positive and direct effect of 0.15 ($z = 2.60$, $p < 0.01$), it indirectly influenced diet-related self-protective response (DSP) through UN. Among the indirect paths, only the path from UN to DSP was significant ($\beta = 0.18$, $z = 2.99$, $p < 0.01$), suggesting that those who felt more uncertain about EDSs risk showed more unlikely to consume riverine fish. UN exerted no mediating role in these relationships, which did not support H1.

3.3.4 Hypothesis 2

It was revealed that the paths from general awareness of water pollution and RHPs (GA), and awareness of the pathways of exposure that may affect reproductive health (AP) to RB ($\beta = 0.25$, $z = 5.19$ and $\beta = 0.16$, $z = 2.95$, respectively), and the path from GA to RC ($\beta = 0.41$, $z = 6.35$) were direct and positive, which supported H2. Whereas, the path from AP to RC was non-significant, which failed to support H2. A post-hoc analysis was conducted for this unsupported direct effect, which revealed a power to detect the non-significant direct effect of AP on RC that might be existing. Therefore, it was confident that this non-significant effect was truly non-significant with the power to detect of 100% with 5% confidence level. The results of H2 are provided in Figure 3.3 and Table 3.6.

3.3.5 Hypothesis 3

Perceived uncertainty (UN) played a partially mediating role in enhancing the relationships of RB and RC with RAC. The effects without- and with- mediation were 0.45 ($p < 0.01$) and 0.56 ($p < 0.01$) for RB, and 0.28 ($p < 0.01$) and 0.44 ($p < 0.01$) for RC, respectively, suggesting that people were more in favor of preventive strategies when they were unsure about EDSs risk. Total effects of RB and RC on RAC were 0.64 ($z = 12.91$, $p < 0.01$) and 0.44 ($z = 9.84$, $p < 0.01$), respectively. UN exerted a full mediation on the path from RC to DSP although the total effect of this relationship was non-significant. These findings, to some extent, provided supporting evidence to H3.

RB successfully explained NDSP with a positive and direct effect of 0.39 ($z = 6.81$, $p < 0.01$), whereas RC presented a non-significant effect on this type of tentative behavior. The relationship of RB with DSP was indirect through UN, where only the indirect path from UN to DSP was significant. UN exerted no mediating role in these relationships, which did not support H3. Within our path model, variance explained for RB, RC, RAC, NDSP, and DSP were 0.24, 0.25, 0.46, 0.24, and 0.04, respectively. The results of H3 are summarized in Figure 3.3 and Table 3.6.

3.3.6 Hypothesis 4

Experience difference

As the results of the chi-square tests, the differences in the paths from risk belief to risk acceptability ($\Delta\chi^2 = 5.0$, $\Delta df = 1$; $p = 0.025$) and from risk belief to NDSP ($\Delta\chi^2 = 4.6$, $\Delta df = 1$; $p = 0.031$) were significant between the experienced and the lay public, which supported H4 (see Table 3.6). The effects of risk belief on risk acceptability and on NDSP were slightly stronger for the experienced public ($\beta = 0.67$ and $\beta = 0.55$) than the lay public ($\beta = 0.53$ and $\beta = 0.29$), respectively.

Pregnancy and child status difference

The chi-square tests revealed a non-significant difference in the hypothesized paths between pregnant women-young mothers and the remaining respondents (see Table 3.6).

3.4 Discussion

3.4.1 Difference between the experienced and the lay public looking from the knowledge aspect

The findings supporting H4 revealed that the patterns of risk belief and their relation to NDSP and risk acceptability differed between the experienced and the lay public. It is argued that lay public tend to make judgments on health and environmental risk from their cognitive heuristics (Slovic 1992, Slovic et al. 1980), whereas experts draw conclusion based on knowledge with assumptions and assessment techniques (Slovic 1997). Compared with those experts participating in risk studies that have been reported, the experienced sample in our research is much more diverse in expertise, however they do share some characteristics in common such as bio-chemical knowledge of life science and/or scientific information on water pollutants and impacts, which distinguish them from the lay public. Therefore, it would be interesting to inspect the difference in the patterns of risk belief between the experienced and the lay public from the knowledge aspect (described in Figure 3.4).

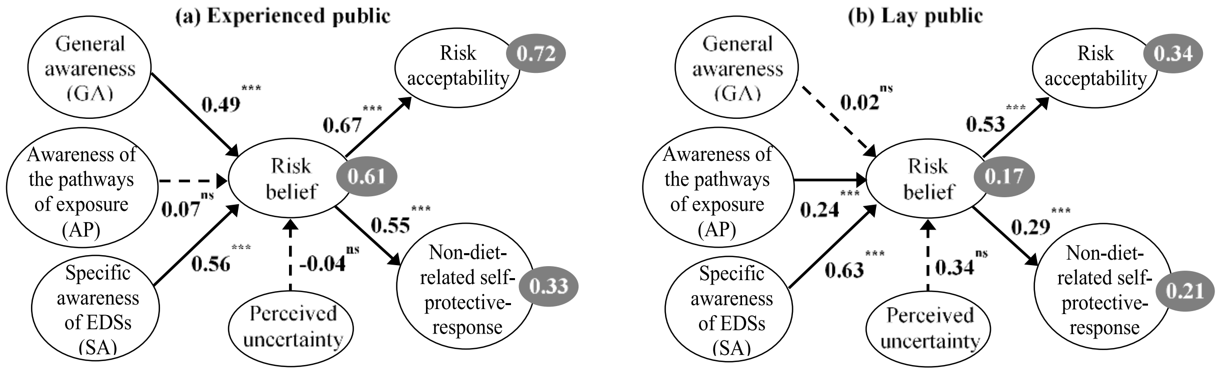

Figure 3.4 Contribution of three levels of knowledge on the risk belief of (a) the experienced and (b) the lay public
Note: *** $p < 0.001$; ^{ns} non-significant.

The multi-sample structural weights model showed that risk belief was attributed to GA much more for the experienced respondents (β = 0.49, $p < 0.01$) than for the opponents (β = 0.02, non-significant), respectively. Experienced people are supposed to have insights of the urban environmental pollution from domestic and industrial activities, which are claimed responsible for EDCs discharge (RPA and BRE 2003, Chau et al. 2015, Duong et al. 2010, Hanh et al. 2014). In an inverse trend, the lay public related AP to EDSs risk more than the opponents did, with corresponding β = 0.24 ($p < 0.01$) and β = 0.07 (non-significant). Surprisingly, SA contributed comparatively to the risk belief of both experienced and lay public with β = 0.56 ($p < 0.01$) and β = 0.63 ($p < 0.01$), respectively. The results seem illogical if the perceived uncertainty is not taken into account. Indeed, it was found that uncertainty exerted a larger effect on risk belief for the lay public (β = 0.34) than for the opponents (β = -0.04) although those direct effects were non-significant. This implies that experts' perceived risk is influenced by knowledge, whereas lay public are probably influenced by their cognitive heuristics. Our finding that three levels of knowledge benefit the experienced and the lay public in different ways deepens the existing theories in an effort to explain ecological and health risk perception in modern society.

The model appeared more parsimonious for the experienced sample to explain risk belief, risk acceptability, and NDSP with variance explained of 0.61, 0.72, and 0.33 than for the inexperienced sample with variance explained of 0.17, 0.34, and 0.21, respectively. Therefore, it is supposed that there are other determinants that could explain EDSs risk belief better for lay public, such as education (Kraus et al. 1992, [EORG] European Opinion Research Group 2002), income (Savage 1993), and media (Frewer 2004). The author also consider about the inequality in sample sizes with 233 inexperienced respondents and 95 experienced respondents.

3.4.2 Perceived uncertainty and cognitive characteristics

In this study, although specific awareness of EDSs (SA) exerted a significant effect on people's risk belief, this effect was weakened by the uncertain feeling. In addition, the respondents showed less concern about the EDSs risk in the presence of uncertainty. The results, on the one hand, are in agreement with scholars in the field of public health who highlight the role of knowledge in explaining risk perception (Agyei-Mensah and Oteng-Ababio 2012, Gaash et al. 2003, Kung and Lee 2006). Noticeably, although the respondents seemed to have moderately low awareness of the EDSs in terms of contamination situation, sources, impacts and pathways of exposure, they showed rather high belief of the chemicals' impacts and concern about this issue. Therefore, it is suggested that the public may have cognitive characteristics that probably affects their perceived risk (Christie et al. 2015, Hertwig et al. 2005). In this situation, cognitive characteristics may be relevant to the following uncertainties: "availability – how humans account for rare events depends upon whether they have experienced them or not", and "anchoring – humans cannot move from preconceptions, but instead anchor to them even in light of new data/information" (Booker et al. 2001). Our introduction about the research as well as the questionnaire itself may stimulate the public to relate EDSs risk with similar preconceptions. This may bring another aspect into the discussion of Pligt (1996) and Maxim et al. (2013) that possible bias in health risk estimation is dependent on risk communication context, risk characterization, as well as personal and cultural characteristics.

On the other hand, the finding that perceived uncertainty due to insufficient risk communication and scientific uncertainty of EDSs impact leads to a decrease in risk perception provides supporting evidence to Funtowicz and Ravetz (1990) and Patt and Schrag (2003). The result is closely connected to the finding that perception of uncertainty is attributed to an inability to precisely identify the causal relationship (e.g., male reproductive disorders due to exposure to EDSs) as well as questions about data, methodology, extrapolation and epistemological validation (Maxim et al. 2013).

3.4.3 Difference in how people take non-diet-related and diet-related self-protective response

The direct and positive effect of risk belief on NDSP is in line with the findings of Lindell and Hwang (2008), Botzen et al. (2009), Siegrist and Gutscher (2006), and Miceli et al. (2008) in environmental hazard domain, and of Sjöberg (2009), Mak and Lai (2012), and Remoundou et al. (2015) in the safety and health protection domain. In spite of that, some studies indicate a minor role of risk perception on precautionary solution (Bubeck et al. 2012). It could be suggested that people are in favor of taking the NDSP as preventive solutions regardless their ambiguous knowledge of EDSs, feeling of uncertainty, and concern about the impacts. In consistency, Heath and Tversky (1991) have found that people with unconscious information tend to show more risk-averse attitudes, and previous works of Vaughan (1993) and Colvin et al. (2013) have provided evidence that self-protective actions are likely taken by those who believe that precautious solutions are helpful.

In contrast, perceived uncertainty was mainly responsible for being unlikely to take DSP, as consuming less riverine fish, despite of high EDSs risk perception. The possibility is that people perceive uncertainty as a barrier and/or they perceive more benefits than reducing exposure. Indeed, scholars have provided evidence that perception of uncertainty is among barriers in decision making, where the uncertainty is related with knowledge (Darlow et al. 2016), experience (Vyner 1988), and consequence of a decision (Chernick et al. 2015, Nardi et al. 2009, Radisic et al. 2017). In addition, people are likely to prioritize the advantageous sides of products (e.g., overall nutrition of fresh food) whilst they perceive deviating from common diet habits in order to mitigate exposure to EDCs as a barrier (Che et al. 2014). Noticeably, the crisis of massive fish deaths along the sea coast of Vietnam that occurred just four months before this survey was conducted significantly limited marine fish supply as well as driving people from consuming seafood to riverine fish. This socio-environmental factor may contribute to explain why people are unlikely to consume less riverine fish.

3.4.4 Contribution to the health belief model

The previous discussions have elucidated the role of specific awareness in explaining risk belief and risk concern, which could be referred to the perceived susceptibility that is described in the health belief model (HBM) (Rosenstock 1974).

In addition, the findings have provided supporting evidence to the mediating role of perceived uncertainty in explaining risk belief, risk concern, and diet-related self-protective response. Perceived uncertainty is referred to the perceived barrier that lays the theoretical foundation for our hypotheses, whilst the tendency of taking self-protective response directly determines one's behavior (Glanz et al. 2008), both of which are the HBM components. The theoretical contribution to the HBM is illustrated in Figure 3.5. In Figure 3.5, shaded boxes and grey lines indicate the constructs and the relationships described in the HBM; unshaded boxes and black lines indicate the modifications by this study. Solid lines refer to direct effects, and dotted lines refer to indirect/ or moderation effects.

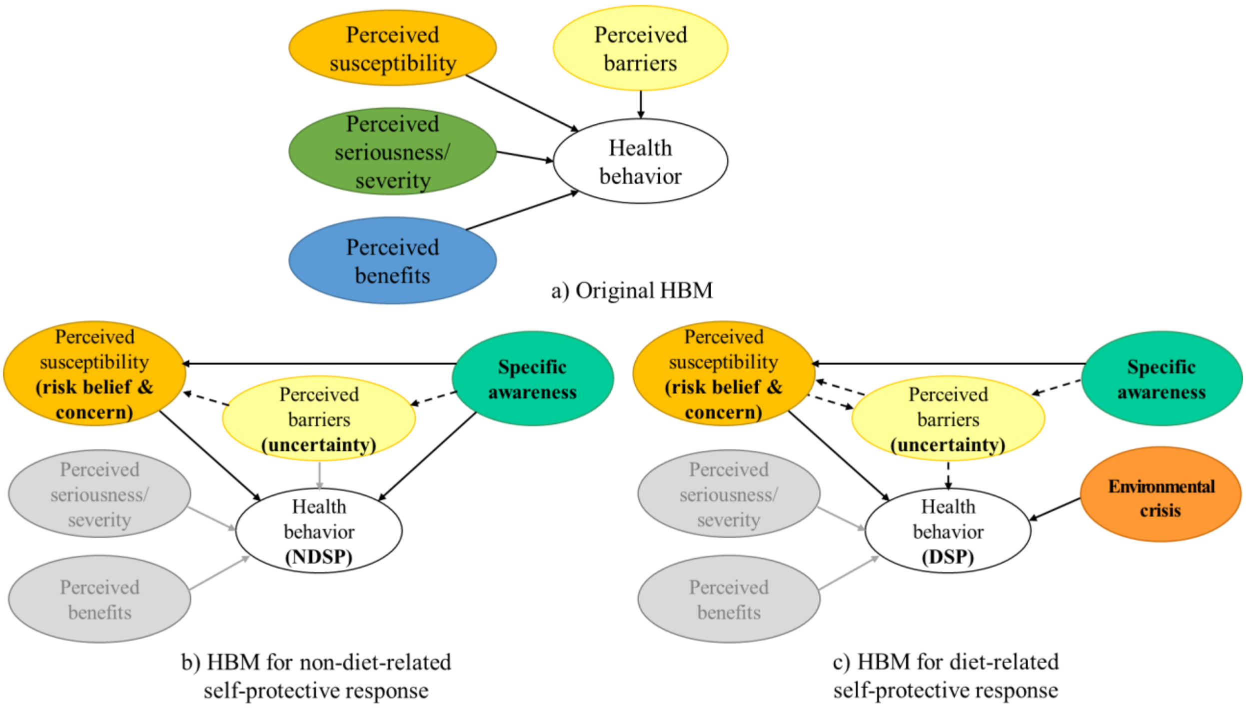

Figure 3.5 Original HBM (Rosenstock 1974) (a) and HBM for non-diet-related self-protective response (b) and for diet-related self-protective response (c) (this study)

3.4.5 Explained variances and potential influencing factors

Within the path model, 24% and 25% of variances of risk belief and risk concern were explained, respectively. Perceiving risk consequences, in reality, is affected by various outrage factors, among which are familiarity, fairness, and dread (Sandman 1987). It follows a pattern that higher perceived risk would be likely with outrages. In this study, the familiarity could be related to an acquaintance with or an experience of RHPs such as abnormalities in reproductive organs, feminization in males, whereas fairness implies an inequality in risk suffering by age and gender (Å. Bergman et al. 2013). Regarding dread, cancers are probably more dreaded than endocrine system malfunction. The three outrages could be potential factors to explain EDSs risk perception. Variance explained for risk acceptability, NDSP, and DSP were 0.46, 0.24, and 0.04, respectively. High responsiveness is related to other factors such as income (Lindell and Hwang 2008), belief in the effectiveness of the measure (Colvin et al. 2013, Vaughan 1993), and perceiving benefits (Che et al. 2014, Rosenstock 1974). Within the scope of this study, the roles of those factors in behavioral decision have not been demonstrated.

3.4.6 Implication for risk communication practice and decision-making process

Communication of EDSs risk would benefit from acknowledging the public's cognitive characteristics regarding EDSs and their dietary habit. The findings suggests that the general public have low awareness of the situation of EDSs contamination, the adverse effect, as well as the exposure routs, which frames the focus of health risk educational programs. The importance of communicating the causal relationship between EDSs exposure and adverse effect is particularly highlighted as suggested by Maxim et al. (2013). Since people are likely to consume riverine fish, detailed dietary advice is needed, for example, which types of fish being ingested at which amounts and frequencies to be considered as no risk. The findings also suggest that health risk education and medical care programs should pay attention on assisting pregnant women-young mothers in early preparation for pregnancy and in baby-care period.

In such situations, building capacity for communicating and managing risk among governmental and non-governmental institutions (e.g., education, research, and medical care organizations) is highlighted. This includes, but not restricted to, EDSs risk-related knowledge and communicating skills that help convey short notices to the public effectively.

3.4.7 Strength and limitation

The strength of this study is that it is based on a validated methodology with reliable measurement scales and constructs, goodness of model fit, considerable R squared, and statistical significance of difference chi-square statistics and individual paths. It is suggested that this model could be replicated for testing other causal paths. In some instances, the author has a confidence that the results could be generalized to urban population in Vietnam.

Nevertheless, this study does have limitation that a 328-respondent sample is not a large sample, so there are deviations from the normality of distributions. Although normal distribution is hardly obtained in social research, it is our responsibility to report for its possible hidden effects on our results. As aforementioned, this study has been carried out in the context of marine environmental crisis in Vietnam. Therefore, the public are supposed to be psychologically affected by this outrage factor. When testing the measurement model, the author has found a significant shared variance of the CLF which reflects a common method bias. This type of CLF could be relevant to "transient mood state" effect, which "refers to the impact of relatively recent mood-inducing events to influence the manner in which respondents view themselves and the world around them" (Podsakoff et al. 2003). Consequently, the CLF may share a common effect of the fear on various environment-related constructs such as GA, AP, risk perception, and acceptability.

3.5 Conclusions

Our findings revealed rather high perception of EDS risk among the respondents, who tended to be in favor of risk mitigation strategies at the governmental administration level and more likely to take non-diet-related preventive measures. The respondents seemed consistent with their diet habits like riverine fish consumption. The patterns of risk belief and their relation to non-diet-related self-protective response and risk acceptability differed by experience among the public. Communicating the societal risk perception and tentative behaviors, one the one hand, the author conveys the public concerning attitudes to relevant stakeholders. On the other hand, the study informatively contributes to and helps enhance the legitimacy of risk decision-making process in Vietnam. As among the few studies in this field using a systematic questionnaire survey and SEM, the study elucidates the roles of three types of knowledge that distinguished by experience, and the mediating role of perceived uncertainty in explaining EDSs risk perception, acceptability, and self-protective response, which have not been mentioned by other scholars.

It is suggested to test this model for other EDCs categories as well as extending the governing factors of risk perception to educational level, income, and media. Additionally, future research should focus on biased media coverage suggested by Slovic et al. (1980) and Lindell and Perry (2003, 1992) in order to explain perceived uncertainty. This is supposed to contribute greatly to EDCs risk communication.

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CHAPTER 4

AN INTEGRATED MODELLING FRAMEWORK FOR EVALUATING NON-CARCINOGENIC HEALTH RISKS FROM NONYLPHENOL CONTAMINATED FOOD CONSUMPTION IN LONG AN, VIETNAM

Summary

This study proposed an integrated modelling framework and a modified method for evaluating non-carcinogenic health risks from nonylphenol (NP) contaminated food consumption. First, a fugacity-based multimedia model and a food web bioaccumulation model were adopted to predict the distribution of NP in the Can Giuoc river and the bioaccumulative concentrations in biota. Next, local people's exposure to NP was quantified using the accumulative concentrations and the data of fishery products intake from a questionnaire survey distributed among 203 local people. Then, human health risk was evaluated in terms of fishery products intake and intake frequency which were each derived from the same survey. The study revealed that human health risk would exist, although the obtained bioaccumulation factors for the consumed organisms were lower than the bioaccumulation criteria. Consuming 141 grams or more per serving of riverine food products resulted in an average NP intake exceeding 0.005 mg per kilogram of body weight per day among 45 - 73% of the local adults, of whom pregnant women or young and potential mothers accounted for 10 – 21%. Seventy-nine percent was the highest rate of the population to be at risk under medium river flow rate when food intake and intake frequency were taken into account. Ingesting 70 grams per serving of more contaminated species, such as whiteleg shrimp and small fish, less frequently could lead to less risk exposure than ingesting 267 grams per serving of less contaminated species, such as sand goby and climbing pearch, more frequently. By coupling food intake with intake frequency, the modified method enables the studying of human health risk from NP contaminated food consumption to be conducted with more care, and so benefits risk communication at local level.

4.1 Introduction

Exposure evaluation is the preliminary step in human health risk characterisation. In order to determine the potency of the non-carcinogenic risk of a chemical, the ratios of exposure levels to a reference dose (RfD) are derived (USEPA 2000). Accordingly, the obtained ratios of greater than one (>1) indicate that the exposed population will be at risk. This method has been widely adopted for evaluating human health risk via oral exposure (e.g., Gyllenhammar et al. 2012, Raecker et al. 2011, Guenther et al. 2002, Lu et al. 2007), where exposure assessment is mainly based on field-measured data. Nevertheless, despite there being great contributions to human health risk assessment, the studies inhere some shortcomings. Firstly, massive field measurements may not be always applicable in some places in the world where access to analytical facilities and finance is restricted. Secondly, intake frequency is not taken into account, although it is noted as one of the factors which influence the probability of adverse effects on a human population (USEPA 2000, Alberto et al. 2015, Thoeye et al. 2003). Concerning the invariable tradeoffs between the protection of public health and the unexpected impacts of consumption restrictions, risk characterisation needs more details (e.g., limit of intake frequency or exposure limit) in order to support risk communication practice. To account for the intake frequency of a multi-species diet, exposure evaluation that is based on the proportions of a given species in an individual's diet (USEPA 2000) seems unreasonable in practice since combinations of species within human diet are diverse. Therefore, this study aims to develop an integrated modelling framework for bioaccumulation assessment under the restricted conditions and to modify the conventional method of non-carcinogenic risk evaluation of USEPA (2000) in a way that integrates intake frequency into the evaluation of health risk from riverine food product consumption.

The study places an interest on nonylphenol (NP) which is known as an endocrine disruptor responsible for adverse effects on the reproductive, immune, and central nervous systems of wildlife and humans (e.g., Vosges et al. 2012, Cosnefroy et al. 2009, Kim et al. 2006, Mao et al. 2010, Razia et al. 2006). It is the most ubiquitous micro-pollutant in the surface water in Vietnam, where the main sources are municipal sewage, industrial wastewater, agricultural runoffs and landfill leachates (Tri et al. 2016). Recent studies have demonstrated the distribution of NP in the waters across Vietnam as well as the accumulation of NP in lower waters that receive wastewater from populous urban areas (Viet et al. 2006, Hanh et al. 2014, Duong et al. 2010, Tam et al. 2016). It is indicated that the aquatic environment of the Can Giuoc river, Long An province, has been deteriorated by the wastewater discharge from the upper urban city. Exposure to NP via riverine food chains may thus pose a health risk to wildlife and the local people. Since the conclusion on the carcinogenic effect of NP has not been fully supported by the available literature ([ECB] European Chemical Bureau 2002), carcinogenic risk assessment will not be under the scope of this study.

4.2 Materials and methods

4.2.1 Study area

Originating from Ho Chi Minh city, the Can Giuoc river connects to the canals in the centre and the south of this city (upper reach), passing Long An province in the east (Can Giuoc and Can Duoc districts), and attributing Soai Rap river at the estuary (lower reach) (Figure 4.1). It is suggested that the river quality has been deteriorated by the municipal wastewater of Ho Chi Minh city (Ha and Phep 2011, Duc et al. 2016). This river serves as the main water source for aquafarming activities and the supply of natural fishery products for the local people in the Can Giuoc – Can Duoc basin.

The Can Giuoc river lies on a flat geography with an elevation of 1-2 m above sea level. This region is characterised by a dry season from December to April and a wet season from May to November when 90% of the precipitation occurs. The temperature is warm and stable with annual averages of 27.0ºC and 27.4ºC in the upper reach and the lower reach, respectively (Phung et al. 2014, SIFEP, 2013).

The river is about 35 km in length and 250 m in average width (measured by Google Earth software). It is influenced by the semi-diurnal tidal regime and saline intrusion from the east sea, thereby characterised by fresh to brackish water. A high salinity of 11 – 25 g/L (2005 - 2011) is observed from January to June, where peak values are reached in March and April (SIFEP, 2013). From July to October, the salinity declines from 12 to 2g/L, and then gradually rises up in November and December.

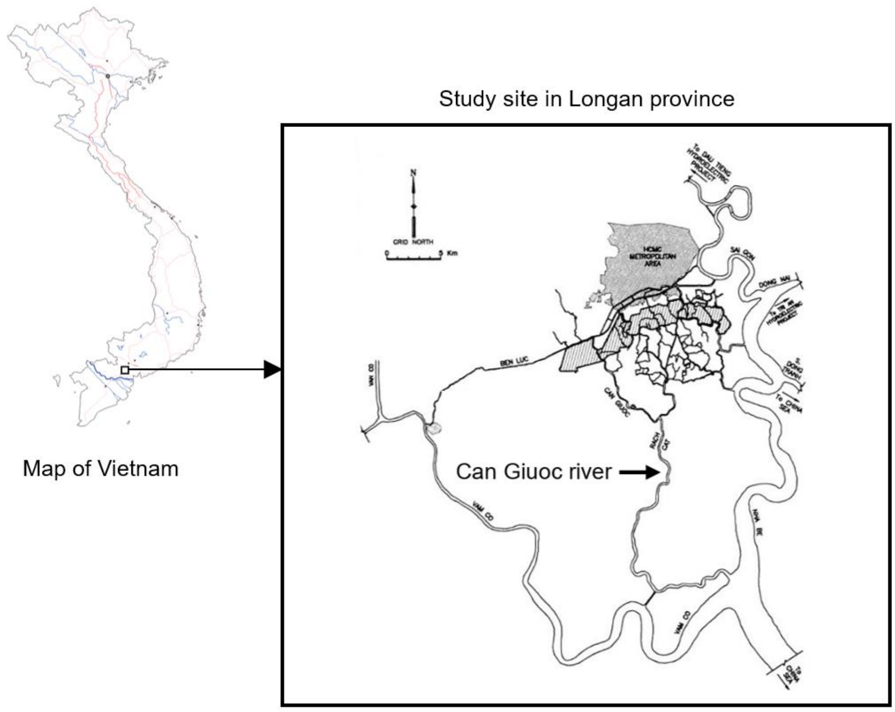

Figure 4.1 The evaluative river basin

4.2.2 Methodological framework

While data regarding the distribution of NP in the studied river as well as the accumulative concentrations of NP in biota was not available, the objectives of the study were fulfilled though a five-stage process (described in Figure 4.2). First, the fate of NP in the Can Giuoc river was predicted using a fugacity-based multimedia model developed by Mackay (2001). Secondly, a food web bioaccumulation model (Arnot and Gobas 2004, Arnot et al. 2008) was adopted to estimate the equilibrium concentrations of NP, the bioaccumulation and biomagnification factors for the species in the Can Giuoc river. Thirdly, human exposure to NP was quantified using the accumulative concentrations in the biota and fishery products intake from a questionnaire survey distributed among the local people. Next, a modified method for human health risk evaluation was proposed on the basis of the current method. Finally, the modified method was applied with data on the frequency of riverine food intake that was derived from the same survey.

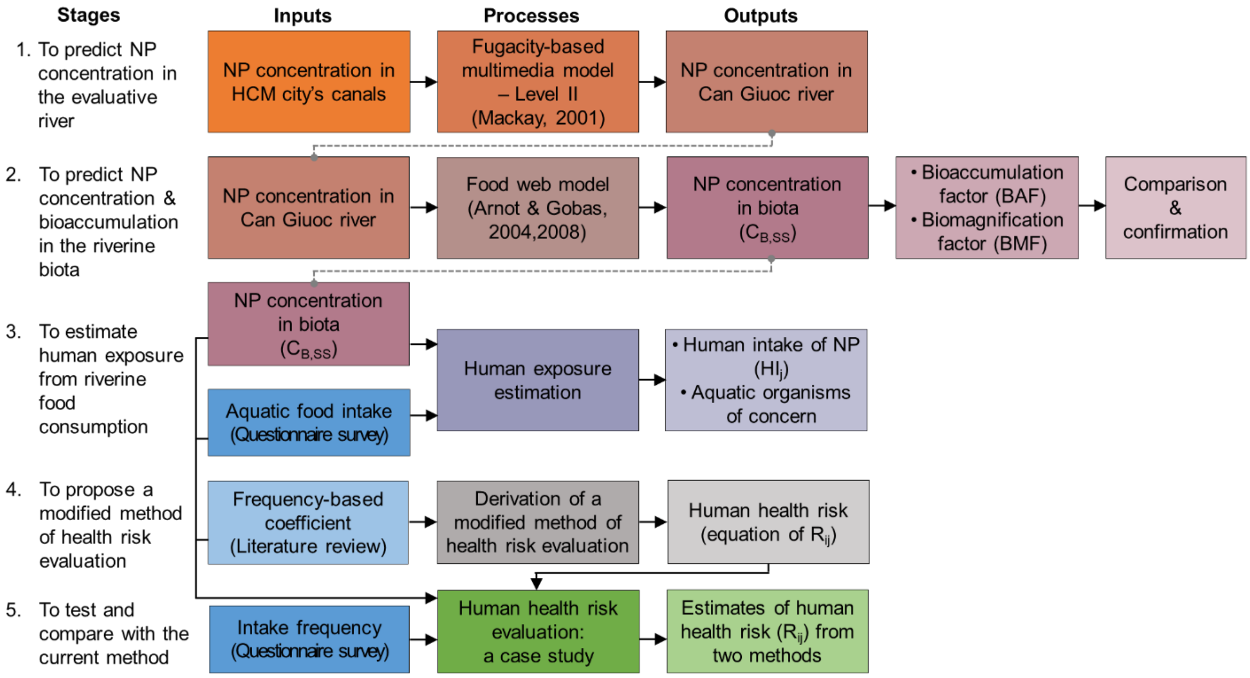

Figure 4.2 Methodological framework

Note: The solid black arrows indicate the flow of data; the dotted grey lines indicate an output to be used as an input in the next step.

4.2.3 Fugacity model

Type of model and assumptions

In this study, a fugacity-based multimedia model (Mackay 2001) was adopted to generate scenarios of NP distribution in a range of site-specific conditions of the Can Giuoc river. Mackay's models have been successfully and widely applied to predict the fate of persistent, bioaccumulative, and toxic substances (PBTs) under environmental risk assessment schemes. There are four basic levels of these models. The multimedia fate model for NP in the Can Giuoc river was level II, which based on some assumptions. The evaluative environment included water, suspended particulate matter (SPM), and sediment compartments, which was considered to be a steady-state system at equilibrium. Located in a rural area, local emission to the evaluative environment was assumed to be zero, and advection into the air was neglected. The evaluative environment's inflow was assumed to equal the outflow of the urban canals. Input NP concentration consisted of dissolved and suspended particulate phases, while transport of surface sediment from the urban canals to the evaluative river was assumed as negligible. Characterized by a high ratio of length over width, it was supposed that the water flow was strictly unidirectional to downstream, and that the water column was completely mixed. The main processes that occurred in the evaluative environment included advection via bulk water, advection via SPM, diffusion between water and sediment, biodegradation in water column, and biodegradation in sediment (Figure 4.3).

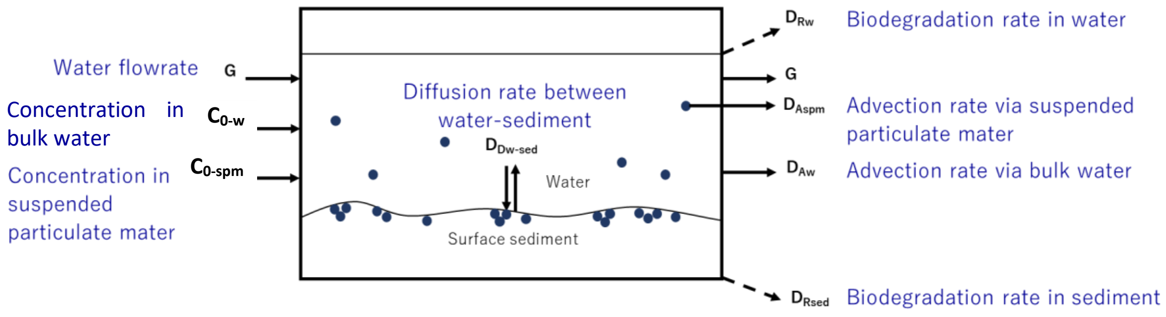

Figure 4.3 The conceptual multimedia fate model for the Can Giuoc river

Mass-balance equations

This model was built on fugacity at equilibrium and mass-balance principles. Fugacity is "identical to partial pressure in ideal gases and is logarithmically related to chemical potential", and thus "linearly or nearly linearly related to concentration" (Mackay 2001). The relationship between fugacity and NP concentration is described by the following equation:

$$\boldsymbol{C_i = Z_i f_i} \qquad \textbf{(1)}$$

where C_i (mol/m^3) denotes the equilibrium concentration of NP, Z_i (mol/m^3.Pa) represents fugacity capacity of NP, and f_i (Pa) indicates the fugacity of NP at equilibrium in the i^{th} environmental compartment.

The overall mass-balance of NP emission flux was as follows:

$$\boldsymbol{N = D_{Aw}f_w + D_{Aspm}f_w + D_{Rw}f_w + D_{Dw\text{-}sed}(f_w - f_{sed}) + D_{Rsed}f_{sed}} \qquad \textbf{(2)}$$

where N (mol/h) denotes total NP emission influx. D_{Aw}, D_{Aspm}, $D_{Dw\text{-}sed}$, D_{Rw}, and D_{Rsed} (mol/h.Pa) represent advection rate via bulk water, advection rate via SPM, diffusion rate between water and surface sediment, biodegradation rate in water, and biodegradation rate in surface sediment, respectively. f_w and f_{sed} (Pa) indicate fugacity in water, and fugacity in SPM and surface sediment, respectively.

The total NP emission influx was estimated using constant rate injection method (De Doncker 2009):

$$\boldsymbol{N = qC_0 = GC_{0\text{-}w} + G_{spm}C_{0\text{-}spm}} \qquad \textbf{(3)}$$

where q, G and G_{spm} (m^3/h) represent a point-source emission flow rate, river flow rate, and the flow rate of SPM flux, respectively. C_0, $C_{0\text{-}w}$, and $C_{0\text{-}spm}$ (mol/m^3) indicate NP concentration in the point-source emission, and in input water as dissolved and suspended particulate phases, respectively. The emission rate would remain unchanged with river flow rates.

Input concentrations of NP in bulk water and surface sediment, and the volatile organic content (VOC) of the sediment were employed in the calculations of initial fugacity values in the input fluxes ($f_{0\text{-}w}$ and $f_{0\text{-}sed}$) (equation 1), as well as organic carbon-normalized partition coefficient *(K_{oc})*. Suspended particulate matter was assumed to own the same characteristics as the surface sediment. The $f_{0\text{-}w}/f_{0\text{-}sed}$ ratio would be maintained in the evaluative environment since the system was at equilibrium. Solving the overall mass-balance of the total NP emission influx (equations 2 and 3), fugacity values in the evaluative environment *(f_w, f_{spm}, and f_{sed})* were derived. On this basis, concentrations of NP at equilibrium and process rates were obtained using equations (1) and (2), respectively.

Details of how to estimate *D, Z* and *f* values as well as the whole calculation process were provided in Appendix 2: Table A2.4 and three scenarios.

Parameterization of the fate model and scenarios

- ***Initial concentrations***

The initial concentration of NP in the dissolved phase of 9.70 µg/L was the average level of NP in Ho Chi Minh city's canals reported by Hanh (2015) who conducted a two-year monitoring programme on organic micro-pollutants in the aquatic environment in Vietnam. K_{oc} value was estimated from average NP concentrations in dissolved and solid (surface sediment) phases, an average VOC of 9.1% (Hanh 2015), and a particulate organic carbon (POC) content of 56% (Mackay 2001). The initial concentration of NP in SPM (3,338 ng/g dw) was derived from the K_{oc} value and an SPM content of 171 g/m^3 (SIFEP, 2013). In the evaluative environment, the K_{oc} would remain the same, whereas the POC content was 4% on average (Minh et al. 2007). The decline in POC is in agreement with Strady et al. (2017) who indicate that the POC in the urban canals' sediment is relatively higher than that in estuaries.

- ***Biodegradation half-life***

Under aerobic conditions, the half-life of NP in water matrix is 20 days at a temperature of 22^{o}C (Staples et al. 1999) or even shorter at about 14 days according to an expert survey (USEPA, 2014). Since the biodegradation of NP is enhanced in warmer temperatures (Ahel et al. 1994, Chang et al. 2004, Manzano et al. 1999, Tanghe et al. 1998, Yuan et al. 2004), a half-life *($t^{w}_{1/2}$)* of 14 days, equivalent to a biodegradation rate *(k_w)* of 0.0021 h^{-1}, was used as a model input.

The half-life of NP in river sediment under aerobic conditions is 20.4 days at 30^{o}C and pH 7, which is shortened to 5.1 days with NP-acclimated sediment (Yuan et al. 2004). The authors also suggest that an addition of salinity (as NaCl) shows inhibitory effects on the biodegradation of NP. Accordingly, a half-life of 49.5 days is achieved at a NaCl concentration of 0.5 mg/L. Under nitrate reducing or anoxic/ facultative conditions, the half-life of NP is 15.1 – 20.1 days (Lu and Gan 2014). In contrast, studies of De Weert et al. (2010), De Weert et al. (2009), Shang et al. (1999) show that NP is highly persistent in sediments, especially under anoxic or reduced conditions. Considering the DO and salinity conditions of the Can Giuoc river, a half-life *($t^{sed}_{1/2}$)* and corresponding biodegradation rate *(k_{sed})* of 49.5 days and 0.0006 h^{-1}, respectively, were selected.

- ***River morphology, hydrological regime, and scenarios***

The hydrological characteristics of the Can Giuoc river were obtained from Mike 11, a modelling software package for rivers and channels developed by DHI Water & Environment, Denmark. The simulation of the river flow rate and velocity from 2010 to 2015 was conducted by the Southern Institute of Water Resources Research of Vietnam in July 2017 by the authors' request under the scope of this study. It revealed that the downward flow dominated the river flow regime over 25 km, from cross-section 1 through cross-section 3 to the river mouth as seen in Figure 4.4. This river section was selected to study the influence of the contaminated water from the upstream city.

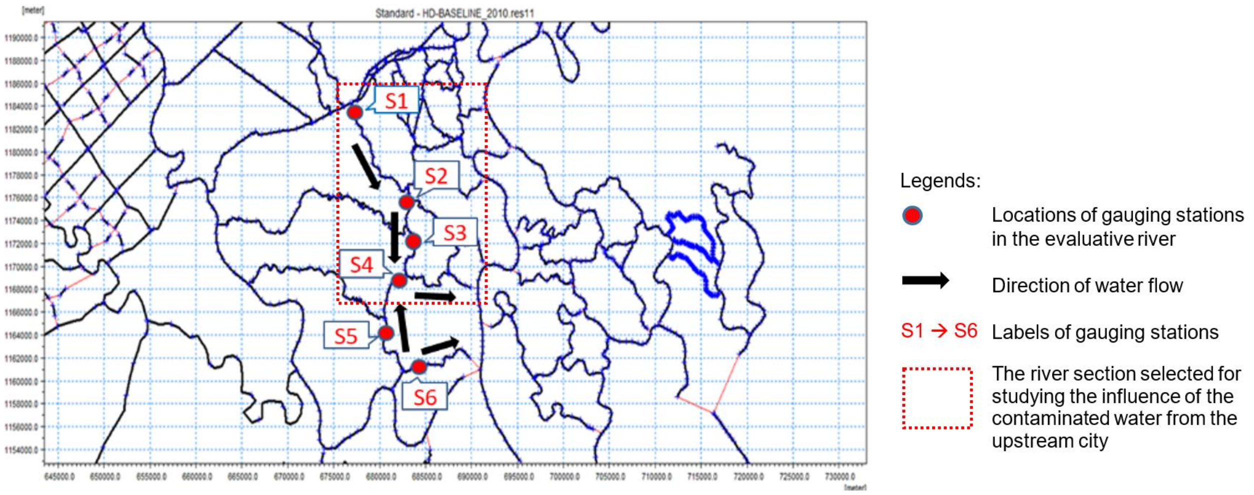

Figure 4.4 Locations of the investigated cross-sections along the Can Giuoc river

The river flow rate is governed by the main factors of precipitation and tidal regime. It ranges from 13.13 (m^3/s) in April to 37.11 (m^3/s) in October with an average value of 20.75 (m^3/s). From these figures, three flow rate-based scenarios were generated for predicting the distribution of NP in the evaluative environment by the aforementioned fate model. Accordingly, max, min, and average flow rates were adopted in high, low, and medium scenarios.

Information on the initial chemical concentrations, the water quality of the Can Giuoc river, the properties of NP, and relevant equations was provided in Appendix 2: Tables A2.1 – A2.3.

4.2.4 Food web bioaccumulation model

Model description

A food web bioaccumulation model (or BAF model) (Arnot and Gobas 2004) that was combined with a biotransformation model (Arnot et al. 2008) was subsequently used for predicting the partitioning of NP into organisms, bioaccumulation and biomagnification factors for each species in the riverine food web. Among the advantages of the integration is that it recognizes the environmental and biological conditions, such as the availability of chemicals, water temperature, and kinetic mechanisms of chemical uptake and elimination in aquatic organisms (Gobas et al.

2009, Arnot and Gobas 2004, Arnot et al. 2008). Being verified with a large empirical data set, the models provide reasonable confidence and convenience for use (Arnot and Gobas 2004).

Parameterization of the food web

Estuary ecosystems are well-known for their species diversity. A riverine food web was developed on the basis of a selection of representative species of all trophic levels that appear abundant in the region, and a literature review along with site-specific knowledge on their habitats, feeding habits, and diet compositions. The resulting food web comprises five compartments: producers (phytoplankton, zooplankton, and macrophyte), primary consumers (worms, gastropod and bivalves, small fish, grass carp, common carp, striped mullet, mullet, tilapia, common shrimp, whiteleg shrimp, and crab), secondary consumers (black seabream, penaied shrimp, and eel), and tertiary consumers (climbing pearch, white flower croaker, bartail flathead, sand goby, spotted snakehead, walking catfish, and yellowfin seabream) and detritus. The characteristics of detritus were based on the river surface sediment. The weight and lipid content of the studied species were literature-derived. Characteristics of the selected species and their trophic interactions were provided in Appendix 3: Tables A3.1 – A3.2.

Trophic levels were recalculated from the trophic interactions among the food web organisms using the following equation (Juntaropakorn and Yakupitiyage 2014):

$$\boldsymbol{TL_{ij} = 1 + \sum_{j=1}^{n} DC_{ij} \times TL_j} \qquad \textbf{(4)}$$

where TL_{ij} represents the trophic level j^{th} of the i^{th} species; DC_{ij} denotes diet compositions that are made up by the ratio of the i^{th} species at the trophic level j^{th} (TL_j).

Parameterization of the food web bioaccumulation model

The outputs of the multimedia fate model, literature-derived site-specific conditions (e.g., water temperature, dissolved oxygen concentration (DO), the concentrations of particulate organic carbon (POC) and dissolved organic carbon (DOC) in water column), the weight and lipid content of the organisms were the inputs of the food web bioaccumulation model. Rate constants of gill uptake (k_1), dietary uptake (k_D), gill elimination (k_2), fecal egestion (k_E), and growth dilution (k_G) were obtained using equations and subordinate rate constants provided by Arnot and Gobas (2004). Biotransformation rates at the original mass and temperature (k_M) for fish species were derived from the average value of whole-body biotransformation rate normalised to a mass of 10g and a temperature of 15°C $(k_{M,N} = 1.7\ d^{-1})$ along with a conversion equation proposed by Arnot et al. (2008). The biotransformation rate of NP by primary producers such as macrophyte and phytoplankton of 0.1 d^{-1} was based on algae *(Cyclotella caspia)* (Liu et al. 2013), whereas the rate for zooplankton and small benthic invertebrates of 10.2 d^{-1} was based on water fleas *(Daphnia magna)* (Preuss et al. 2008). The biotransformation rates by worms were estimated from the difference between the elimination rates of 0.251 d^{-1} *(Perinereis nuntia)* (Nurulnadia et al. 2013) and 0.192 d^{-1} *(Lumbriculus variegatus)* (Maenpaa and Kukkonen 2006) and the predicted egestion rate constants and growth dilution rate constants. The obtained values for mud worm and earth worm were 0.235 and 0.151 d^{-1}, respectively. Similarly, estimates of the biotransformation rates by shrimps and crab were based on the elimination rate of *Gammarus pulex* of 0.24 d^{-1} (Gross-Sorokin et al. 2003). The obtained values for common shrimp, penaied shrimp, whiteleg shrimp, and crab were 0.214, 0.138, 0.227, and 0.217 d^{-1}, respectively.

The bioaccumulation factor (BAF – L/kg) and biomagnification factor (BMF - dimensionless) were derived from the following equations (Arnot and Gobas 2006):

$$\boldsymbol{BAF = \frac{C_{B,SS}}{C_{WT,O}}} \quad \mathbf{(5)}$$

$$\boldsymbol{BMF = \frac{C_{B,SS}}{C_{D}}} \quad \mathbf{(6)}$$

where $C_{B,SS}$ (g/kg ww) denotes the equilibrium concentration in biota at steady state, $C_{WT,O}$ (g/L) indicates the total chemical concentration in bulk water phase, and C_D (g/kg ww) is the equilibrium concentration in prey. A BMF value of greater than one (>1) indicates biomagnification of a chemical in a species.

All parameters and equations of the food web accumulation model for the Can Giuoc river were provided in Appendix 3: Food web bioaccumulation model and Tables A3.3 – A3.5.

4.2.5 Questionnaire survey

The residents in the Can Giuoc river basin include those who locate in Can Giuoc and part of Can Duoc districts. Local people of at least 18 years old, of all genders, were subjected to a questionnaire survey in September 2017. They were met at their homes which were randomly selected. Socio-demographic information such as age, sex, status of child, and status of pregnancy was derived by closed-ended questions. Information on fishery products intake and intake frequency of riverine food products that comprised eighteen species was obtained from semi open-ended questions. The fishery products intake scale included four options: (1) if possible, specify an average amount in gr/person.serving; (2) <100 gr/person.serving; (3) 100 – 200 gr/person.serving; (4) >300 gr/person.serving. Questions on intake frequency were modified in more details based on Sise and Uguz (2017). Accordingly, three levels were adopted in the intake frequency scale: (1) very frequently (almost everyday); (2) frequently (the participants were required to state how many times per week); and (3) sometimes (the participants were required to state how many times per month). Data on the intake frequency was subsequently homogenised into time(s) per month for further calculations. Questionnaires were pretested with 20 people to guarantee their transparency and understandability. After a minor revision, the questionnaires were self-administered by 203 local people.

4.2.6 Chemical intake

Chemical intake by humans was quantified using the following equation:

$$\boldsymbol{HI_j = I_j \times C_{B,Ave} \div W_{H,Ave}} \quad \textbf{(7)}$$

where HI_j (mg/kg bw. serving) represents human intake of NP at the j^{th} level of fishery products intake; I_j (g/serving) denotes the j^{th} level of fishery products intake; $C_{B,Ave}$ (g/kg_ww) indicates the average wet weight concentration of NP in biota; and $W_{H,Ave}$ (kg) is average human body weight. The average body weight of Vietnamese adults is 53.33 kg (Walpole et al. 2012).

4.2.7 Human health risk evaluation - A modified method

Non-carcinogenic risk for human health is determined using a ratio of exposure to the RfD *(R = HI/RfD)*, which measures how much exposure deviates from the RfD (USEPA 2000). The tolerable daily intake (TDI) value of 0.005 mg of NP per kilogram of body weight per day is used as the RfD in many cases (Lu et al. 2007, Osimitz et al. 2015, Huang et al. 2014, e.g., Ademollo et al. 2008). The TDI value for NP was derived by the Danish Institute of Safety and Toxicology (Nielsen et al. 2000). It was calculated from a lowest-observed-adverse-effect level (LOAEL) of 15 mg/kg.day via oral exposure and three safety factors (SF_I, SF_{II}, and SF_{III}). The value was based on histopathological kidney changes in rats and minor perturbations in the reproductive system of rat offspring. SF_I was set to 10 assuming that humans were more sensitive than animals, an SF_{II} of 10 was used to protect the most sensitive individuals in the population, and SF_{III} was set to 30 because the derivation was LOAEL-based and data for genotoxicity and carcinogenicity were insufficient. The selection of the safety factors to some extent agrees with Duah (1998).

It is suggested that the reference dose used in the hazard-index-based method of risk characterisation does not truly reflect the toxicity of the chemical (Sarigiannis and Hansen 2012). This is because the derivation of reference dose includes uncertainty factors that are not fully data-based but also assumption-based. The magnitude of an uncertainty factor depends on the reliability of the corresponding N(L)OAEL value, which is governed by the number of species and trophic groups involved in relevant dose-response tests. As such, the TDI value for NP inheres both types of uncertainty due to data shortage and assumptions, suggesting an overestimation of risk. Solving this knowledge gap requires further toxicological studies.

By integrating intake frequency, this study introduces an alternative approach to contribute to tackling the overestimating problem of the current method. The traditional method may also include intake frequency rather than exposure. Nevertheless, the frequency is on a regular basis by default; hence, the magnitude of risk associated with varied intake frequency cannot be realised. In reality, research has recognized the positive associations of the frequency of fish intake and using cleansing products with the level of NP in breast milk (Huang et al. 2014, Chen et al. 2010, Sise and Uguz 2017). This may relate to a rapid absorption from the gastrointestinal tract and an extensive metabolism via glucuronide and sulphate conjugation (Müller et al. 1998). In particular, the level of NP in breast milk of mothers who consumed fish and used cleaning products at least once or twice per week was approximately twofold higher than that of those consuming fish and using cleaning products only once or twice per month or seldom (Sise and Uguz 2017). This provides evidence that fish intake frequency does influence human exposure to NP. The literature lays a foundation for the proposal of a modified equation to evaluate non-carcinogenic human health risk from NP contaminated fish consumption as follows:

$$\boldsymbol{R_{ij} = \frac{F_i HI_{ij} n}{RfD}} \tag{8}$$

where R_{ij} (dimensionless) and HI_{ij} (mg/kg bw. serving) denote human health risk and human intake of NP from ingesting the i^{th} organism in the food web at the j^{th} amount. HI_{ij} is derived using equation (7). F_i (dimensionless) indicates the frequency-based coefficient corresponding to the i^{th} organism. n represents the number of servings per day for the same type of food (organism), and RfD (mg/kg bw. day) denotes a reference dose.

The derivation of the frequency-based coefficient was based on the findings of Sise and Uguz (2017) as follows (Table 4.1).

Table 4.1 Frequency of fish intake and frequency-based coefficient

Frequency of fish intake	Frequency-based coefficient (F_i)
At least 4-8 times/month	1.0
3 times/month	0.75
1-2 time(s)/month or seldom	0.5

4.3 Results and discussion

4.3.1 Distribution of nonylphenol in the Can Giuoc river

The results of the fugacity-based fate model showed that the concentrations of NP in both dissolved and solid phases declined when the water coming from the upper canals flowed through the territory of Long An province. The equilibrium concentrations of NP were 4.86 (μg/L) in water and 1,324 (ng/g dw) in SPM and surface sediment for low scenario (11.13 m^3/s), which decreased to 3.09 (μg/L) and 842 (ng/g dw) in the corresponding compartments for high scenario (37.11 m^3/s) (Table 4.2). In all scenarios, the dissolved concentrations were much greater than the European Union environmental quality standard of 0.33 μg/L (European Parliament and Council 2008).

The mass-balance-based calculations revealed that advection via bulk water and biodegradation in bulk water were two main mechanisms that governed the distribution of NP in the evaluative river. While the former process was enhanced at higher water flow rates, the latter process was diminished under the same conditions. Being greatly influenced by advection processes at elevated water flow rates, the overall residence time of NP in the evaluative environment was shortened from 21.0 days to 13.3 days under low and high scenarios, respectively (Table 4.2).

Table 4.2 Distribution of NP in the Can Giuoc river, process rates, and residence time of the water under three scenarios

Parameter	Scenario		
	Low	Medium	High
River flow rate G (m^3/s)	13.13	20.75	37.11
Initial concentration in water $C_{0\text{-}w}$ (µg/L)	15.34	9.70	5.43
Initial concentration in SPM & surface sediment $C_{0\text{-}spm/sed}$ (ng/g dw)	5,279	3,338	1,868
Equilibrium concentration in water* C_w (µg/L)	4.86	4.11	3.09
Equilibrium concentration in SPM & surface sediment* $C_{spm/sed}$ (ng/g dw)	1,324	1,120	842
Advection via bulk water* $D_{Aw}f_w$ (mol/h)	1.04	1.39	1.87
Advection via bulk SPM* $D_{Aspm}f_w$ (mol/h)	0.05	0.06	0.09
Diffusion between water & surface sediment* $D_{Dw\text{-}sed}(f_w - f_{sed})$ (mol/h)	0.18	0.15	0.12
Biodegradation in bulk water* $D_{Rw}f_w$ (mol/h)	1.69	1.43	1.08
Biodegradation in sediment* $D_{Rsed}f_{sed}$ (mol/h)	0.51	0.44	0.33
Overall residence time* τ (day)	21.0	17.7	13.3

Note: * Results of the fugacity-based multimedia model.

The estimated equilibrium concentrations are lower by factors of 1.7 – 3.1 than the initial concentrations at the point source. This is comparative with the field-based data of Shao et al. (2005) who report decreases in concentration by factors of 1.2 and 2.2 in the Yangtze and Jialing rivers, respectively, and of Zhang et al. (2011b) who document a decrease by a factor of 1.7 over 50 km in the Jialu river. It is suggested that the concentration of NP is higher in high tide than in ebb tide (Yang et al. 2016), probably because of the accumulation of wastewater discharge during high tide. A shortcoming of the multimedia fate model is that it simplified the situation with the net flow rate of the river, where the influence of tidal phases on NP concentration variation was neglected.

4.3.2 Bioaccumulation in the food web organisms

Equilibrium concentrations of NP in biota

The results showed a decrease in the concentrations of NP in the organisms based on both wet weight (muscle for fish and eel; soft tissue for invertebrates; whole body for the other species) and lipid weight at higher river flow rates. The concentrations ranged from $3.5x10^{-4}$ to $1.7x10^{-2}$ (g/kg ww) and from $1.9x10^4$ to $4.2x10^5$ (ng/g lw) on wet weight and lipid weight basis, respectively (Table 4.3). NP is a comparative lipophilic compound (Ahel and Giger 1993), which explains higher NP concentrations (wet weight based) in organisms of higher lipid content. Good examples are phytoplankton compared with macrophyte, and whiteleg shrimp with crab, which are quite similar in feeding habits.

Diet preference does govern chemical concentration in each species (Burkhard 2003). As the results of the estimation, NP concentrations in worms ranked at top levels, followed by whiteleg shrimp (see Appendix 4: Figure A4.1). This is because detritus, as a sink of NP, predominates in the diet of worms (as primary consumers), which are further consumed by secondary consumers such as whiteleg shrimp and crab. The study suggests that sediment plays an important role in the bioaccumulation of NP in organisms that feed on detritus and bottom dwellers, which is in agreement with the findings of Greenfield et al. (2015), Gross-Sorokin et al. (2003), and Nurulnadia et al. (2013).

Lipid content influences the lipid-normalized concentration in an adverse pattern. The lipid-normalized concentrations of NP of macrophyte and benthic invertebrates were higher than that of other species (see Appendix 4: Figure A4.2). Indeed, the lipid contents of macrophyte and benthic invertebrates were lowest among the studied species, at 0.02% and 0.01%, respectively.

Bioaccumulation factors and biomagnification factors

The estimation of the bioaccumulation factor revealed wet weight bioaccumulation factor (BAF_ww) values to be lower than the bioaccumulation criteria (<5000) (Government of Canada 1999), and lipid-normalized bioaccumulation factor (BAF_lw) values to be greater than the bioaccumulation criteria (>5000) for all species (Table 4.3). It is suggested that chemicals that do not meet the criteria for persistence, bioaccumulation, and toxicity are not subjected to risk assessment unless there is information that motivates this process (Gobas et al. 2009). However, people feed on fish not only for proteins and micronutrients, but also for essential fats (Kawarazuka and Béné 2011). Considering the high values of BAF_lw among the species and the food consumption patterns of the local people, there deserves to be a health risk assessment conducted in the case of NP.

The relatively low BAF_ww values obtained in this study are comparable with the BAFs for benthic invertebrates such as oyster *(Crassostrea gigas)* and mussel *(Mytilus californianus)*, and fish such as arrow goby *(Clevelandia ios)* from the North American Pacific Coast estuaries reported by Diehl et al. (2012). However, the results are two orders of magnitude greater than the BAFs for phytoplankton, zooplankton, mussel, and fish (e.g., flounder, herring, cod) estimated by Staniszewska et al. (2014), probably because those values are based on a single prey. Indeed, BAF values of one food web can differ from another food web due to many factors such as the temperature, environmental concentrations, organic matter, trophic structure and interaction (Gobas and MacLean 2003, Burkhard 2003).

Species with the lipid-normalized BMF of greater than one (>1) included those feeding on detritus (e.g., worms, whiteleg shrimp, and crab), and those relying mainly on planktons (e.g., benthic invertebrates, common shrimp) and worms (e.g., eel, penaied shrimp), which had a trophic level of between 2 and 3. The estimated BMF values for the mud worm (3.6) and earth worm (4.4) were highest and greater than the field-based BMF for lugworm *(Arenicola marina)* (Korsman et al. 2015) a factor of 47 and 57, respectively. The BMFs for whiteleg shrimp, common shrimp, and penaied shrimp were also 2 – 10 times higher than the field-based values for brown shrimp *(Crangon crangon,* BMF = 0.5*)* (Korsman et al. 2015) and ghost shrimp *(Neotrypaea californiensis,* BMF = 0.2*)* (Diehl et al. 2012). NP was magnified in benthic invertebrates 37 times greater than in oyster *(Crassostrea gigas)* as measured by Diehl et al. (2012). The BMFs for fish species ranged from 0.1 to 0.9, which were comparative to the reported values for arrow goby *(Clevelandia ios,* BMF = 0.3*)* (Diehl et al. 2012), but lower than those for sole *(Solea solea)* feeding on brown shrimp (*Crangon crangon)* (BMF = 1.8), and feeding on sprat *(Sprattus sprattus)* (BMF = 7.9) (Korsman et al. 2015). The difference in predicted and field-based BMF values has been showed to contribute to several factors, such as feeding interaction, site-specific characteristics (e.g., latitude, longitude), and environmental conditions (e.g., nutrition, organic carbon, temperature) (Radomyski et al. 2018, Nurulnadia et al. 2014, Magali et al. 2008). As aforementioned, since bioaccumulation studies assume single predator-prey pairs, their estimated BMFs vary in comparison with the results of this study, which assumes relative complicated trophic interactions. In addition, the biotransformation rate for fish is enhanced in elevated temperatures (this study) (Arnot et al. 2008), thus explaining why the BMFs for fish species are lower than the reported values.

A linear regression investigation on the lipid-normalized BMF values and the trophic levels under three scenarios revealed a statistically significant declination of BMF at higher trophic levels ($p < 0.05$), suggesting a trophic dilution in the investigated food web (see Appendix 4: Figure A4.3). Accordingly, an increase of one trophic level would reduce the BMF value by a factor of 1.29, and this relationship was statistically significant. Since humans directly consume biomagnified species such as whiteleg shrimp, penaied shrimp and eel which have trophic levels of (i), it is assumed that the trophic level for humans is ($i + 1$). The BMF values for humans based on a single prey of whiteleg shrimp, penaied shrimp, and eel would be 0.7, 0.0, and 0.0, respectively, suggesting that NP is not biomagnified in humans via the examined food web.

Table 4.3 Chemical concentration in biota under three scenarios, bioaccumulation factors, and biomagnification factors

Species	Scientific name	Trophic level	Whole body weight W_B (kg)	Lipid fraction f_{LB} (g/g)	Wet weight equilibrium concentration in biota $C_{B,SS}$ (g/kg ww)			BAF_ww (L/kg)	Lipid-normalized equilibrium concentration in biota $C_{B,SS}$ (ng/g lw)			BAF_lw (L/kg)	BMF (-)
					Low scenario	Medium scenario	High scenario		Low scenario	Medium scenario	High scenario		
Macrophyte	*Myriophyllum spicatum*	1.0	-	0.002[a]	7.6E-04	6.5E-04	4.9E-04	150.2 - 151.8	3.8E+05	3.2E+05	2.5E+05	75,106.8 - 75,923.9	-
Phytoplankton	na	1.0	-	0.023[b]	3.5E-03	3.0E-03	2.3E-03	688.3 - 695.8	1.6E+05	1.3E+05	1.0E+05	30,592.6 - 30,925.5	-
Zooplankton	na	2.0	5.70E-08[c]	0.018[b]	3.3E-03	2.8E-03	2.1E-03	652.9 - 660.0	1.8E+05	1.5E+05	1.2E+05	35,482.8 - 35,868.8	1.2
Whiteleg shrimp	*Litopenaeus vannamei*	2.0	0.016[d]	0.050[d]	6.3E-03	5.3E-03	4.0E-03	1,237.5 - 1,250.1	1.3E+05	1.1E+05	8.1E+04	24,749.4 - 25,001.8	2.0
Mud worm	*Perinereis nuntia*	2.0	0.001[e]	0.134[f]	1.6E-02	1.4E-02	1.0E-02	3,124.6 - 3,156.6	1.2E+05	1.0E+05	7.6E+04	23,318.2 - 23,556.9	3.6
Earth worm	*Lumbriculus variegatus*	2.0	6.00E-06[g]	0.122[h]	1.7E-02	1.5E-02	1.1E-02	3,424.7 - 3,459.9	1.4E+05	1.2E+05	9.2E+04	28,186.6 - 28,476.8	4.4
Grass carp	*Ctenopharyngodon idella*	2.0	1.000[i]	0.016[j]	1.5E-03	1.2E-03	9.4E-04	286.0 - 289.1	9.4E+04	8.0E+04	6.0E+04	18,451.1 - 18,651.8	0.2
Crab	*Sesarma dehaani*	2.0	0.020[k]	0.020[k]	3.4E-03	2.9E-03	2.2E-03	664.9 - 671.6	1.7E+05	1.4E+05	1.1E+05	32,914.7 - 33,249.3	3.7
Mullet	*Liza soiuy*	2.0	0.093[b]	0.044[b]	2.3E-03	2.0E-03	1.5E-03	452.4 - 457.1	5.3E+04	4.5E+04	3.4E+04	10,375.6 - 10,483.0	0.1
Striped/grey mullet	*Mulgil cephalus*	2.1	0.379[l]	0.051[m]	2.3E-03	1.9E-03	1.4E-03	443.6 - 448.1	4.5E+04	3.8E+04	2.9E+04	8,749.2 - 8,838.7	0.4
Benthic invertebrates	(e.g. *Assiminea lutea, Corbicula lutea*)	2.1	0.005[i]	0.001[n]	5.4E-04	4.6E-04	3.5E-04	106.7 - 107.8	4.2E+05	3.5E+05	2.7E+05	82,100.9 - 82,945.9	3.7
Tilapia	*Oreochromis niloticus*	2.1	0.330[d]	0.030[d]	2.0E-03	1.7E-03	1.3E-03	396.3 - 400.6	6.7E+04	5.7E+04	4.3E+04	13,211.2 - 13,354.9	0.4
Small fish (as prey)		2.3	0.005[i]	0.034[i]	2.6E-03	2.2E-03	1.6E-03	502.8 - 508.3	7.4E+04	6.3E+04	4.8E+04	14,608.7 - 14,767.0	0.4
Common shrimp	*Gammarus pulex*	2.6	0.050[o]	0.014[p]	2.6E-03	2.2E-03	1.7E-03	519.5 - 524.8	1.9E+05	1.6E+05	1.2E+05	37,104.3 - 37,485.0	1.6
Common carp	*Cyprinus carpio*	2.8	0.330[j]	0.022[j]	1.7E-03	1.5E-03	1.1E-03	339.2 - 342.7	7.8E+04	6.6E+04	5.0E+04	15,348.7 - 15,505.3	0.9
Sand goby	*Glossogobius giuris*	2.9	0.106[q]	0.013[r]	1.6E-03	1.4E-03	1.1E-03	322.2 - 325.5	1.2E+05	1.0E+05	7.8E+04	23,955.8 - 24,202.2	0.7
Eel	*Monopterus albus*	3.0	0.288[l]	0.007[m]	1.3E-03	1.1E-03	8.2E-04	250.4 - 253.0	1.9E+05	1.6E+05	1.2E+05	37,939.3 - 38,329.1	1.1
Penaied shrimp	*Metapenaeus affinis*	3.0	0.015[s]	0.013[s]	2.9E-03	2.5E-03	1.9E-03	568.4 - 574.3	2.2E+05	1.9E+05	1.4E+05	43,721.9 - 44,173.5	1.2
Back seabream	*Acanthopagrus schlegelii*	3.1	0.300[i]	0.067[t]	2.8E-03	2.3E-03	1.8E-03	543.3 - 548.9	4.1E+04	3.5E+04	2.6E+04	8,072.7 - 8,156.1	0.1
Climbing pearch	*Anabas testudineus*	3.1	0.070[u]	0.012[u]	1.5E-03	1.3E-03	9.9E-04	303.4 - 306.7	1.3E+05	1.1E+05	8.3E+04	25,496.3 - 25,773.3	0.6
Walking catfish	*Clarias batrachus*	3.1	0.330[i]	0.036[v]	2.2E-03	1.9E-03	1.4E-03	438.0 - 442.6	6.2E+04	5.3E+04	4.0E+04	12,168.0 - 12,293.6	0.5
Yellowfin seabream	*Sparus latus*	3.1	0.315[w]	0.082[x]	2.5E-03	2.1E-03	1.6E-03	487.7 - 492.7	3.0E+04	2.6E+04	1.9E+04	5,947.4 - 6,008.7	0.1
White flower croaker	*Nibea albiflora*	3.2	0.195[b]	0.031[b]	2.2E-03	1.9E-03	1.4E-03	432.1 - 436.6	7.0E+04	5.9E+04	4.5E+04	13,761.2 - 13,903.4	0.4
Bartail flathead	*Platycephalus indicus*	3.2	0.357[b]	0.025[b]	2.0E-03	1.7E-03	1.3E-03	395.6 - 399.7	7.9E+04	6.7E+04	5.1E+04	15,575.4 - 15,736.4	0.4
Spotted snakehead	*Channa maculata*	3.3	1.500[m]	0.018[m]	1.7E-03	1.4E-03	1.1E-03	329.3 - 332.7	9.3E+04	7.9E+04	5.9E+04	18,192.7 - 18,380.6	0.5

Notes: BAF_ww and BAF_lw are wet weight and lipid-normalised accumulation factors, respectively. BMF is based on lipid-normalised equilibrium concentration in organisms (ng/g lw) and organic carbon-normalised equilibrium concentration (ng/g OC) in detritus.

Sources: [a] Gobas et al. (1991); [b] Hu et al. (2005); [c] Arnot and Gobas (2004); [d] Rico (2014); [e] Honda and Kikuchi (2002); [f] Limsuwatthanathamrong et al. (2012); [g] Cowgill et al. (1986); [h] Kukkonen and Landrum (1994); [i] estimated by the authors; [j] Manpreet Kaur (2010); [k] Gao et al. (2009); [l] Wei et al. (2011); [m] Cheung et al. (2007); [n] Diehl et al. (2012); [o] Sutcliffe et al. (1981); [p] Nyman et al. (2013); [q] Tuan et al. (2014), [r] Islam and Joadder (2005); [s] Dincer and Aydin (2014); [t] Xuan et al. (2013); [u] Sanatan et al. (2016); [v] Minh et al. (2006); [w] Phu and Dao (2013); [x] Zhang et al. (2011a).

4.3.3 Human intake of nonylphenol

The data on fishery products intake and intake frequency was derived from a questionnaire survey conducted among 203 participants. The sample was dominated by young (ages 18 – 40) and middle (ages 41 – 60) adults (94%), females (67%), and those who have no child or whose child's age was not less than six (83%). Among the females, 58% were non-pregnant, and 27% were planning for pregnancy or being pregnant.

About 27% of the participants who were asked about fishery products intake could provide numerical responses on a wide range from 50 to 300 g/serving with an average of 162 g/serving. The remaining respondents provided the fishery products intake in a three-interval scale, of which 19% were less than 100 grams per serving, 33% were in between 100 and 200 g/serving, and 20% were greater than 200 g/serving. The numerical responses were categorized into the three-interval scale and the average fishery products intake in correspondence with each interval was calculated as 70, 141, and 267 g/serving.

The data of fishery products intake, in addition to the average equilibrium concentration of NP obtained from 19 species in the food web (except for macrophyte, planktons, worms, and common shrimp), was used for the estimation of the chemical intake from riverine food consumption under three scenarios: low, medium, and high flow rates using equation (7). As a result, the chemical intake by the local public was in the ranges of 0.003 – 0.011, 0.003 – 0.010, and 0.002 – 0.007 mg NP/kg bw.serving, which appeared to decline slightly under low, medium and high scenarios, respectively (Table 4.4). The results are about 3.4 – 19 – fold and 3.8 – 21 – fold higher than those of Lu et al. (2007) and Niu et al. (2015), who report the average NP intakes of being 31.4 µg/day among Taiwanese and 520 ng/kg bw.day among Chinese adults, respectively, from consuming various types of foods.

Under the low scenario, approximately 73% of the population who consumed a considerable amount of 141 g or more per serving would definitely have the NP intake of greater than the tolerable daily intake (TDI) of 0.005 mg NP/kg bw.day. Under the medium and high scenarios, about 28% of the population would have the NP intake of exceeding the TDI value. The rate of having NP intake that exceeded the TDI for pregnant women or young and potential mothers was 10 – 21% of the population.

Table 4.4 Chemical intake in correspondence with riverine food intake and the percentage of population

Fishery products intake – I_j (g/serving)	Percentage of population (%)	Percentage of pregnant women, young & potential mothers (%)	Chemical intake – HI_j (mg/kg bw. serving)			Tolerable daily intake – *TDI* (mg/kg bw. day)
			Low scenario	Medium scenario	High scenario	
70	26	8.4	0.003	0.003	0.002	0.005
141	44	10	0.006	0.005	0.004	
267	28	11	0.011	0.010	0.007	

Whiteleg shrimp and crab appeared to pose health risks to consumers under all scenarios, whereas small fish, penaied shrimp, and seabreams should be of concern under the low and medium scenarios (see Appendix 4: Figure A4.4). Benthic invertebrates (e.g., small bivalves and gastropod), carps, eel, climbing pearch, sand goby, and spotted snakehead (edible muscles) could be regarded as safe food products under all scenarios.

4.3.4 Human health risk evaluation

A demonstrative case study in the Can Giuoc river basin under the medium scenario was conducted by applying the modified method of human health risk evaluation. The data for the estimation was derived from the aforementioned questionnaire survey. Accordingly, the average fishery products intakes of 70, 141, and 267 g/serving were adopted. The average intake frequency for each species in time(s) per month was derived for three food-intake categories and then transferred into frequency-based coefficient indexes using Table 4.1. The intake frequency of less than once per month was considered to be seldom. It appeared that people who had larger portions of fishery products (in terms of serving size) were likely to consume fishery products more frequently (Table 4.5). In addition, tilapia, sand goby, and climbing pearch were found to be eaten most frequently, whereas white flower croaker and seabreams were the least preferred types.

Next, the frequency-based coefficient indexes for the organisms in the riverine food web were adopted to estimate human health risk using equation (8). It was assumed that the number of servings for each organism was once a day (n = 1). The TDI value of 0.005 mg NP/kg bw.day was used as the RfD in this case. The comparison of human health risk from consuming riverine food products derived by the current and the modified methods was shown in Table 4.6.

Table 4.5 Intake frequency and frequency-based weighting coefficient for each species

i	Organism	I_1 = 70 g/serving		I_2 = 141 g/serving		I_3 = 267 g/serving	
		Frequency[a] (times per month)	Frequency-based coefficient (F_i)	Frequency[b] (times per month)	Frequency-based coefficient (F_i)	Frequency[c] (times per month)	Frequency-based coefficient (F_i)
1	Whiteleg shrimp	1	0.5	0.5	0.5	4	1.0
2	Penaied shrimp	2	0.5	4	1.0	5	1.0
3	Small fish	1	0.5	2	0.5	5	1.0
4	Walking catfish	1	0.5	3	0.75	5	1.0
5	Tilapia	3	0.75	5	1.0	12	1.0
6	Common carp	1	0.5	1	0.5	4	1.0
7	Spotted snakehead	2	0.5	4	1.0	4	1.0
8	Sand goby	2	0.5	4	1.0	11	1.0
9	Climbing pearch	2	0.5	4	1.0	10	1.0
10	Mullet	-	-	1	0.5	1	0.5
11	Striped/ grey mullet	0.2	0.5	0.5	0.5	0.2	0.5
12	White flower croaker	0.02	0.5	0.5	0.5	0.04	0.5
13	Bartail flathead	0.1	0.5	0.3	0.5	0.2	0.5
14	Grass carp	0.2	0.5	0.2	0.5	1	0.5
15	Eel	0.3	0.5	0.2	0.5	1	0.5
16	Benthic invertebrates	2	0.5	3	0.75	3	0.75
17	Back seabream	0.02	0.5	0.1	0.5	-	-
18	Yellowfin seabream	0.02	0.5	0.1	0.5	-	-

Notes: [a, b, c] Average values based on 53, 91, and 57 respondents, respectively. "-" data not available.

Table 4.6 Estimates of human health risk associated with each organism by the current and the modified methods

i	Organism	Estimates of human health risk by the current method ($R_{ij} = HI_{ij}/TDI$)			Estimates of human health risk by the modified method ($R_{ij} = F_iHI_{ij}n/TDI$)		
		I_1 = 70 g/serving	I_2 = 141 g/serving	I_3 = 267 g/serving	I_1 = 70 g/serving	I_2 = 141 g/serving	I_3 = 267 g/serving
1	Whiteleg shrimp	1.4	2.8	5.3	0.7	1.4	5.3
2	Penaied shrimp	0.6	1.3	2.5	0.3	1.3	2.5
3	Small fish	0.6	1.1	2.2	0.3	0.6	2.2
4	Walking catfish	0.5	1.0	1.9	0.2	0.7	1.9
5	Tilapia	0.4	0.9	1.7	0.3	0.9	1.7
6	Common carp	0.4	0.8	1.5	0.2	0.4	1.5
7	Spotted snakehead	0.4	0.8	1.4	0.2	0.8	1.4
8	Sand goby	0.4	0.7	1.4	0.2	0.7	1.4
9	Climbing pearch	0.3	0.7	1.3	0.2	0.7	1.3
10	Mullet	0.5	1.0	2.0	-	0.5	1.0
11	Striped/ grey mullet	0.5	1.0	1.9	0.3	0.5	1.0
12	White flower croaker	0.5	1.0	1.9	0.2	0.5	0.9
13	Bartail flathead	0.4	0.9	1.7	0.2	0.5	0.9
14	Grass carp	0.3	0.7	1.2	0.2	0.3	0.6
15	Eel	0.3	0.6	1.1	0.1	0.3	0.5
16	Benthic invertebrates	0.1	0.2	0.5	0.1	0.2	0.3
17	Back seabream	0.6	1.2	2.3	0.3	0.6	-
18	Yellowfin seabream	0.6	1.1	2.1	0.3	0.6	-

The conventional health risk evaluation revealed that ingesting 267 g/serving of most species, except for benthic invertebrates (small gastropod and bivalves), would pose a health risk linked to NP towards the local community as the R_{ij} values were greater than one ($R_{ij} > 1$). Those consuming smaller amounts of whiteleg shrimp (70 g or more per serving), penaied shrimp, small fish, and seabreams (141 g/serving) would also be at risk, whereas ingesting other species was considered to be no risk. The fishery products are consumed on a regular basis, therefore the magnitude of health risk is associated with the sole factor of food intake.

In contrast, the results of the modified method indicated that ingesting 267 g/serving of some species, such as mullets, white flower croaker, bartail flathead, grass carp, and eel, would pose no health risk to the local community, which was not obtained by the current method. It also revealed that less frequently consuming a small amount (70 g/serving) of a more contaminated species, such as whiteleg shrimp and small fish, could result in an individual being exposed to a lower health risk than if they more frequently consumed a large amount (267 g/serving) of a less contaminated species, such as sand goby and climbing pearch. The magnitude of health risk from consuming 267 g/serving of the most preferred species (e.g., tilapia, sand goby, climbing pearch, and penaied shrimp) obtained equal values from both methods, which fit the assumption regarding the regular intake of the current method. When integrating frequency-based coefficients, the magnitude of health risk from certain species reflects the associations of two dietary patterns: intake amount and intake frequency.

The percentage of the population at risk was obtained from both methods and compared. Individual data of intake frequency and average fishery products intakes of 70, 141, and 267 g/serving were then adopted. The intake frequency of each organism was also converted to a frequency-based coefficient using Table 4.1. The highest health risk value for each individual was selected. The percentages of people at risk among the sample population were then calculated for three food-intake categories and for pregnant women or young and potential mothers within each category. The conventional method revealed 99% of the local people to be at risk as the R_{ij} values exceeded one ($R_{ij} > 1$), whereas the rate derived from the modified method was 79% (Table 4.7). Similarly, the rates of pregnant women or young and potential mothers to be at risk were 30% and 24% by the conventional method and the modified method, respectively. The difference in the percentages of the population at risk from the two methods is obviously due to the dis-/integration of the intake frequency in the human health risk evaluation. In particular, smaller rates obtained from the modified method were observed for the low food-intake category (I_1 = 70 g/serving). This is consistent with the aforementioned finding that people who have smaller portions of fishery products are likely to consume fishery products less frequently, hence being exposed to lower health risk. The findings suggest that the conventional method overestimates the health risk from riverine food consumption more than the modified method does.

Table 4.7 Percentage of the population at risk due to riverine food consumption

Human health risk ($R_{ij} > 1$)	Estimates by the current method				Estimates by the modified method			
	$I_1 = 70$ g/serving	$I_2 = 141$ g/serving	$I_3 = 267$ g/serving	Total	$I_1 = 70$ g/serving	$I_2 = 141$ g/serving	$I_3 = 267$ g/serving	Total
Percentage of the sample (%)	26	45	28	99	6.4	45	28	79
Percentage of pregnant women, young & potential mothers (%)	8.4	10	11	30	3.0	10	11	24

On the other hand, the conventional method allows for the inference of health risk (R_{ij}) directly by comparing chemical intake (HI_{ij}) with TDI value. Nevertheless, this process may not be applicable for the modified method since it considers another factor that also influences heath risk, the intake frequency. In this case, low intake frequency may yield a lower health risk. Indeed, the HI_{ij} values obtained from both methods were equal, yet the R_{ij} values and the percentages of the population at risk were lower for the modified method.

4.3.5 Sensitivity of the models

It is suggested that the multimedia fate model for NP is sensitive to river water discharge *(G)*, organic carbon-normalized partition coefficient *(K_{oc})*, and the biodegradation rate in water column *(k_w)* (Zhang et al. 2011b), whereas food web bioaccumulation model seems the most sensitive to biotransformation rate of biota *(k_M)* (Mackintosh et al. 2004, Korsman et al. 2015, Arnot and Gobas 2006). Linking to K_{oc}, organic carbon content has a strong influence on bioaccumulation factor variation (Greenfield et al. 2015). In practice, the river flow rate varies by months, and many factors may affect the measurement of organic carbon fraction, k_w, and k_M. Since humans could be direct predators of the riverine organisms, variation in environmental conditions is supposed to impact chemical intake *(HI_j)* and health risk *(R_{ij})* via food consumption. Being included in the estimation of HI_j and R_{ij}, the influence of food intake variation was not subjected to a sensitivity test. Hence, the sensitivities of NP concentrations in the evaluative environment and biota, bioaccumulation factor, biomagnification factor, chemical uptake, and human health risk were each investigated by comparing the medium scenario (as a baseline) with others where each of the following variables: G, POC, k_w, and k_M were changed by +/-10%. A summary of the sensitivity results was presented in Table 4.8. The sensitivity index was as follows: a change of no less than (≥) 1% indicated high sensitivity; a change of no less than (≥) 0.1% indicated moderate sensitivity; a change of less than (<) 0.1% indicated low sensitivity, and a 0.00% change indicated no sensitivity (Zhang et al. 2011b).

The results showed that the concentrations of NP in the environment and biota were highly sensitive to G, POC, and k_w, where POC exerted a greater influence over the NP concentrations in SPM and sediment with their changes of +11.8/-11.6%. Bioaccumulation factor on both wet weight and lipid weight basis demonstrated a moderate sensitivity to the three variables, where greater changes (-0.7/+0.7%) were also observed in the variance of POC. The biomagnification factor revealed low sensitivity to insensitivity for most species when G and k_w varied by +/-10%. However, it exhibited a moderate sensitivity for worms and striped mullet when POC varied by +/-10%. In agreement with the literature, k_M showed great impact on the concentrations in biota and the bioaccumulation factor (changes of -7.5/+8.8%), and the biomagnification factor (changes of -6.6/+7.7%) for most species of investigation. Both human uptake of NP and the estimates of human health risk by the modified method were highly sensitive to the variance in all variables, among which the variance in POC showed weaker impact (changes of +1.5/-1.6%), whereas the variance in k_M had the strongest influence on the estimates of human health risk for most species (changes of -7.5/+8.8%). This suggests that human health risk varies with the metabolic capacity of the consumed organisms that are influenced by environmental conditions and the growth stage.

Table 4.8 Results of sensitivity analyses

<table>
<tr><td>Model</td><td colspan="3">Fugacity-based fate model</td><td>Food web model</td></tr>
<tr><td>Variable</td><td>River flow rate
G (m^3/s)</td><td>Particulate organic carbon
POC (%)</td><td>Biodegradation rate in water
k_W (1/h)</td><td>Biotransformation rate
k_M (1/d)</td></tr>
<tr><td>Calibrated value</td><td>20.75</td><td>4.0</td><td>0.0021</td><td>for all species</td></tr>
<tr><td>Variance (%)</td><td>+/- 10</td><td>+/- 10</td><td>+/- 10</td><td>+/- 10</td></tr>
<tr><td>NP concentration in water - C_W (µg/L)</td><td rowspan="2">Highly sensitive (+5.6/-6.1)</td><td>Highly sensitive (+1.6/-1.8)</td><td rowspan="2">Highly sensitive (-4.0/+4.3)</td><td rowspan="2">-</td></tr>
<tr><td>NP concentration in SPM & sediment $C_{SPM/sed}$ (ng/g dw)</td><td>Highly sensitive (+11.8/-11.6)</td></tr>
<tr><td>NP concentration in biota C_{B_WW} (g/kg ww) and C_{B_LW} (ng/g lw)</td><td>Highly sensitive (+5.4/-5.9)</td><td>Highly sensitive (+1.6/-1.7)</td><td>Highly sensitive (-3.9/+4.2)</td><td rowspan="2">Highly sensitive (up to -7.5/+8.8) for most species, except for planktons, earthworm, and penaied shrimp (moderately sensitive; up to -0.6/+0.7), and macrophyte (low sensitive)</td></tr>
<tr><td>Bioaccumulation factor BAF_ww (L/kg) and BAF_lw (L/kg)</td><td>Moderately sensitive (-0.1/+0.2)</td><td>Moderately sensitive (-0.7/+0.7)</td><td>Moderately sensitive (+0.1/-0.1)</td></tr>
<tr><td>Biomagnification factor BMF (-)</td><td>Insensitive to low sensitive for most species, except for mud worm and earth worm (moderately sensitive, -0.1/+0.1)</td><td>Insensitive to low sensitive for most species, except for mud worm and earth worm (moderately sensitive, -0.2/+0.2), and striped mullet (moderately sensitive, +0.1/-0.1)</td><td>Insensitive to low sensitive</td><td>Highly sensitive (-6.6/+7.7) for most species, except for zooplankton, earth worm, common shrimp, and eel (moderately sensitive, -0.7/+0.7)</td></tr>
<tr><td>Chemical uptake by human HI_j (mg/kg bw. serving)</td><td>Highly sensitive (+5.4/-5.9)</td><td>Highly sensitive (+1.4/-1.6)</td><td>Highly sensitive (-3.9/+4.2)</td><td>Highly sensitive (-4.6/+5.2)</td></tr>
<tr><td>Human health risk derived from the modified method R_{ij} (-)</td><td>Highly sensitive (+5.4/-5.9)</td><td>Highly sensitive (+1.5/-1.6)</td><td>Highly sensitive (-3.9/+4.2)</td><td>Highly sensitive (-7.5/+8.8), except for penaied shrimp (moderately sensitive, -0.6/+0.7)</td></tr>
</table>

Notes: HI_j was based on an average fishery products intake and an average equilibrium concentration of NP obtained from 19 species in the food web (except for macrophyte, planktons, worms, and common shrimp). R_{ij} incorporated three levels of fishery products intake, the equilibrium concentration of NP for each consumed species, and the intake frequency for each species obtained from the survey. R_{ij} includes HI_{ij} – chemical intake for a certain species consumed at a certain level, which varies in the same order as R_{ij} when G, POC, k_w, and k_M changes.

4.3.6 Practical and policy implications

The methodological framework of the study could be suggested for screening ecological risk and health risk of other chemicals and useful in places where there is a lack of field-measured data. The modified method advances the health risk characterization process by coupling food intake with intake frequency, which allows to evaluate non-carcinogenic human health risk with more care and to compare different dietary patterns. The details on risk exposure among the local adults and the species of consideration in the local diet are essential for risk communication. Since food consumption patterns may vary by areas due to geographic characteristics, the outputs of the modified method would benefit risk communication at city or provincial level.

Based on the findings of health risk from consuming riverine food products (this study) and high risk perception (study 1), it is suggested actions in order to tackle this issue need a kick start in urban cities. Revised standards for sewage and industrial effluent discharge and for surface water that integrate NP/NPEOs are needed. This will require assessments and monitoring of wastewater treatment facilities with regards to EDSs removal as a short-term strategy. Long-term strategies should aim to raise the awareness of health risk among the public as well as to restrict the use of EDSs among industrial manufactures. In addition, the moral and legal responsibilities of the "polluter" towards the "sufferer" of the pollution should be recognized in the legal system of environmental protection of Vietnam.

4.4 Conclusion

The study provides the insight into NP bioaccumulation in a tropical estuary food web while such studies are rare. The findings indicate that human health risk will exist, although the obtained BAF values for the consumed organisms are below the bioaccumulative criteria. The study provides original information on the exposure to NP and health risk via riverine food consumption among the neighbouring community to the polluted city of Ho Chi Minh. Accordingly, 45 - 73% of the local adults, among whom 10 - 21% are pregnant women or young and potential mothers, have the average NP intake of an excess of 0.005 mg/kg bw.day. Seventy-nine percent is the highest rate of the population to be at risk under the medium river flow rate when food intake and intake frequency are taken into account.

Since this is the first study to address human health risk from NP contaminated riverine food consumption, further studies should extend the focus to other types of food product and drinking water. At this stage NP is the only among EDCs found in the diet of local people that affects human health, therefore further research on human health risk from exposure to mixtures of EDCs is highlighted. In addition, the water pollution in Ho Chi Minh city has been demonstrated to influence the ecosystem of the Can Giuoc river and the health of the local people who rely on the river. This suggests further studies to consolidate the ecological impact on not only this river but also other rivers in urban cities.

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CHAPTER 5

A FIRM BEHAVIOR MODEL BASED ON A QUALITATIVE ANALYSIS OF THE VIETNAMESE TEXTILE FIRM'S RESPONSE TO THE RESTRICTION OF NONYLPHENOL AND NONYLPHENOL ETHOXYLATES

Summary

Since textile industry has been claimed to endanger the environment and human health, the insight of textile firm behavior regarding chemical use and discharge is vital for designing environmental risk management strategies. This study aimed to explore the dynamics of responses to the restriction of nonylphenol (NP) and its ethoxylates (NPEOs) among the Vietnamese textile manufacturers from the perspectives of attitude and the perceptions of adaptabilities, risks, benefits, and barriers. The chemicals are used as surfactants and known to be responsible for endocrine disrupting effects. In-depth interviews were conducted with technical specialists from four textile firms and one chemical supplier. Regulatory and market situations with regards to the chemicals were also assessed. The findings revealed varied responses to chemical elimination where perceived technical risk, financial risk, benefits, and barriers played different roles in driving a certain action. The attitude towards chemical restriction was shaped by the trade-off between perceptions of financial risk and benefits and was moderated by market strategy. Efforts, such as enhanced washing or reductions in the dose of NP/NPEOs, imply the potency of continuous discharge of these chemicals into the environment, suggesting critical investigations on NP/NPEOs removal to prioritize actions for balancing between economic growth and environmental protection. Poor access to new policies and technological and chemical innovations was the most important barrier among private firms, highlighting the roles of non-governmental textile and garment industrial/ trade associations in enhancing their members' informative capacity. The study reflects the significance of incorporation of firm behavior research into environmental risk management practice.

5.1 Introduction

Nonylphenol (NP) and its ethoxylates (NPEOs) have been detected in textile products in many countries of manufacture and use, providing evidence that the chemicals are commonly used within transnational clothing supply chains (Brigden et al. 2012, Brigden et al. 2013). The chemicals are released into the environment during the manufacturing processes and use of textile products, mainly as wastewater discharge (Pothitou and Voutsa 2008, Ho and Watanabe 2017). As the final decomposed metabolite of NPEOs, NP is known to have endocrine disrupting effects on the reproductive, immune, and central nervous systems of wildlife and humans (Kim et al. 2006, Razia et al. 2006, Vosges et al. 2012).

Concerning the adverse effects of NP/NPEOs, in 1998 both chemicals were added to the list of chemicals needing priority action in order to protect the marine environment of the North-East Atlantic (OSPAR 1998). Their use has been regulated under the Registration, Evaluation, Authorization and Restriction of Chemicals (REACH) program of the European Commission since 2006 (EC 2006). Under this scheme, after 3 February 2021, if the residual level of the chemicals in textile articles is equal or greater than 0.01% by weight such articles may not be placed on the European market (EC 2016). Other countries such as Taiwan, Korea, and the United States have also launched or proposed safety requirements regarding the residues of NP/NPEOs in textile products with a focus on infant and child health protection (AAFA 2017, BSMI 2013, KATS 2016).

At present, NP/NPEOs are still permissible for use in Vietnam. This not only poses local ecological and human health risks but also emerges as a barrier for exporting textile goods to foreign markets due to the regulations on the residual levels in textile products. Therefore, eliminating the use and discharge of NP/NPEOs is crucial. This arouses a question as to how the Vietnamese textile manufacturers respond to the need of NP/NPEOs restriction in their manufacturing processes and products. The responses are expected to be diverse, ranging from relying on the state's regulatory enforcement to conforming to international regulations or adopting mixed patterns. In the current context of Vietnam, conformance to international regulations could be regarded as the decision of taking voluntary actions which may involve creative problem solving (Russo and Fouts 1997).

Existing literature supporting the research question studies the influences of external factors such as legal, social, and economic pressures (Hoffman 2001, Gunningham et al. 2004, 2003, Liu et al. 2010, Singh et al. 2014), as well as internal factors such as resources and strategies (Aragón-Correa et al. 2008, Christmann and Taylor 2006, Galende and de la Fuente 2003) on firm behavior and performance. Notwithstanding, it is argued that individual perspectives such as managerial attitude, value, and the perceptions of costs, risks, and benefits play salient roles in driving firm managers to take a certain action (e.g., Egri and Herman 2000, Nakamura et al. 2001, Sharma 2000, Wu 2009). Perceived organizational and technological adaptabilities are also found to impact firm innovative behavior and performance (Meeus and Oerlemans 2000). Noticeably, most of scholars study the effect on single environmental decision in terms of regulatory compliance or violation or voluntary strategy, not on a decision of mixed patterns that may be a case in this study. Given that the choice of environmental strategy is governed by external factors (regulatory and market pressures) and internal factors (attitude and perceived organizational and technological adaptabilities, risks, benefits, and barriers), the dynamics of how these factors shape diverse responses are not well understood. The knowledge serves as a foundation for proposing and compiling multiple actions into an effective national policy for ecological and human health protection as well as reinforcing the textile exporting capacity of Vietnam.

The study objective is threefold. First, the tentative behavior of textile firms will be examined through technical specialists' perceptions of organizational and technological adaptabilities, risks, benefits, and barriers, as well as their attitude. Second, environmental regulations and market situation regarding NP/NPEOs will be assessed in order to provide the insight of the context within which the textile firms operate. The third objective is to propose a conceptual model for examining mechanisms of behavioral diversity.

5.2 Materials and methods

5.2.1 Formulation of conceptual model

Various theories have been used to explain firm behavior. Among those are the "license framework" (Gunningham et al. 2004, 2003), the institution theory (Hoffman 2001, Powell and DiMaggio 2012, Scott 2003), the theory of rational choice (Paternoster and Simpson 1996, 1993), the evolutionary theory of the firm (Teece and Pisano 1994, Haveman 1992), and the combination the approaches (Delmas and Toffel 2004, Rorie 2015, Meeus and Oerlemans 2000, Testa et al. 2016).

The "license framework" of Gunningham et al. (2004, 2003) explains corporate behavior that is constrained by external factors in three domains: the "legal license", the "socio license", and the "economic license". Accordingly, the legal license includes the pressures from regulatory enforcement and criminal justice sanction; the social license refers to the external pressures from social institutions such as media, politics, public, and non-governmental organizations; and the economic license indicates the ability of financial gain or toleration as the behavioral consequence. To some extent, although sharing some common features with the "license framework", institutional studies of organizations distinguish themselves by placing the focus on the interaction mechanisms between organizations and the external environment. In other words, studies have analyzed the effects of external pressures that are filtered through the interpretation of managerial individuals, as well as the collective interpretation by organizational participants through three major attributes: culture-cognition, values and norms, and legal and rule-based system (Scott 2003).

The literature on firm behavior describes diverse patterns of interaction. It has been suggested that through coercive, mimetic, and normative isomorphism mechanisms, firms operating in a certain field tend to adopt similar practices and to develop similar structures so as to respond to the external environment (DiMaggio and Powell 1983). In contrast, rationalist scholars argue that firms operating in the same environment may appear in heterogeneous behaviors, depending on the value, the perceived cost, risks and benefits, and the attitude of managerial individuals, who are driven by competition and efficiency (e.g., Egri and Herman 2000, Nakamura et al. 2001, Sharma 2000, Wu 2009).

Previous empirical studies suggest that some factors influence both environmental overcompliance and violation, whilst other factors could explain only overcompliance or violation. Legal penalties (Nyborg and Telle 2006, Karpoff et al. 2005), frequent inspections (Rousseau 2008, Earnhart 2004), market competition, and firm characteristics (e.g. small, publicly traded, and intermediate firms) (Wu 2009) are found to deter environmental violations. Interestingly, voluntary pollution reductions and overcompliance with environmental standards are mostly driven by regulatory pressure (e.g., Innes and Sam 2008, Videras and Alberini 2000), and personal values and beliefs of upper management (e.g., Wu and Wirkkala 2009). In the association with both types of behavior, costs and risks of undertaking environmentally friendly practices and technologies are found to enhance environmental violations and thus discourage overcompliance (Wu 2009). Multiple dimensions of risks (e.g., financial, operational, strategic) (Kenett and Raanan 2011) and of benefits (e.g., profit, competitiveness, reputation) (Wu 2009) have been recognized.

The restriction on the use and discharge of NP/NPEOs as well as the residual levels in textile articles may require technical interventions in terms of chemicals, manufacturing processes, and probably upgrading wastewater treatment facilities. Regarding this, firm behavior could be viewed from an adaptive perspective derived from the evolutionary theory (Haveman 1992, Teece and Pisano 1994). The analysis of firm's adaptabilities towards change is based on a framework composed of three domains of a firm: strategy, process, and structure that support adaptation to change (Miles et al. 1978). Meeus and Oerlemans (2000) regard these domains as strategic, technical, and organizational adaptations, respectively, which have been proved to have positive effects on the innovative performance of small-sized firms. Daniel and Amos (2013) suggest that the influence is through manager's perceptions of risk and benefit as monetary loss and gain, respectively.

It is not known how the Vietnamese textile firms perceive and internalize the external pressures regarding the restriction of NP/NPEOs from legal and market drivers. The knowledge could be referred to firm's attitude and perceived adaptabilities along with perceived risks, benefits, and barriers to undertaking necessary technical interventions. In order to operationalize the research aim, a conceptual model of firm behavior (Figure 5.1) was formulated based on existing theories. The model is composed of two external factors (regulatory enforcement and market pressure) and four internal factors as perception (1 – technical and organizational adaptabilities; 2 – risks in terms of manufacturing process, product quality, and cost; 3 – benefits regarding profit, market opportunity, and competitiveness; and 4 – barriers) that were hypothesized as influential to a firm's attitude[2] and tentative behaviors[3].

[2] Attitude refers to the way a person views, feels about, or behaves to (e.g., a concern about/ a regard for/ an interest in) the applicability of the foreign regulation.

[3] Tentative behavior indicates the response that firms may take to align with the foreign regulation in the future.

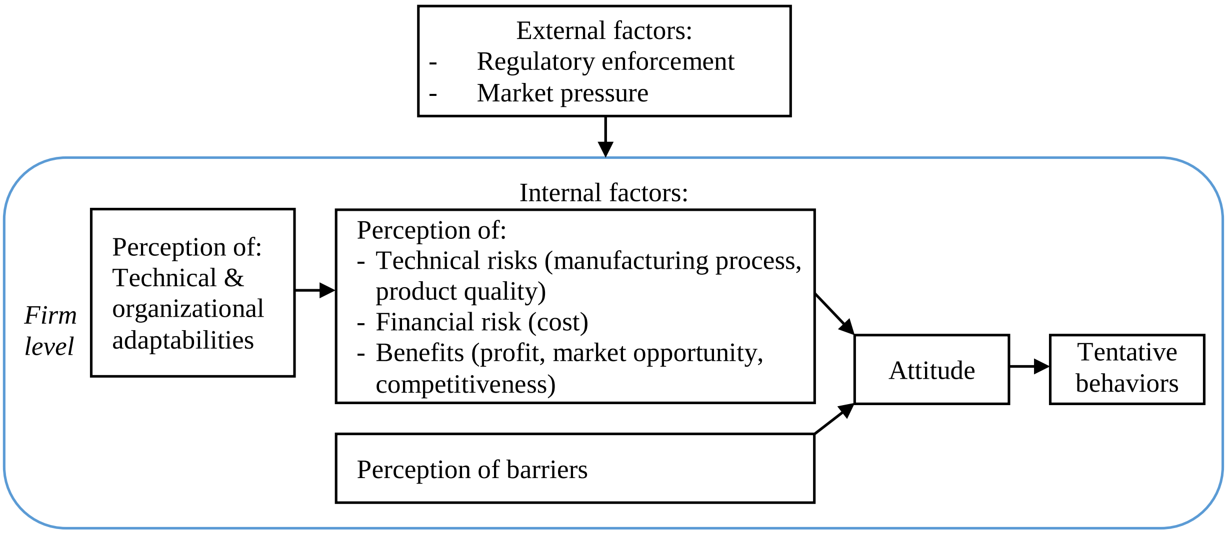

Figure 5.1 A conceptual model of firm behavior

5.2.2 Data collection

Answering the research question was mainly based on semi-structured in-depth interviews, a widely used qualitative method (Barbara and Benjamin 2006). The method is considered to be appropriate in a sense that it provides "a greater understanding of individuals' reactions", thus "leads to unexpected results, and allows us to investigate complex phenomena" (Stiles et al. 2006). The interviews were conducted with five technical specialists working in the textile industry (four with minimum 10-year experience, and one with minimum 5-year experience) who are responsible for manufacturing processes or technical functions of their factories and giving advisory opinions to managerial level. It is suggested that individuals in an organization focus on different aspects of the organization, and how they view the world depends on their cognitive frame (Hoffman 2001). Since this research focuses on issues related to chemical use and substitution, the technical specialists are supposed to be appropriate interview candidates.

The technical specialists came from four textile firms (two small-medium-sized firms and two large-sized firms), and one transnational chemical supplier (Table 5.1). The interviews were arranged at the respondent's offices in September 2017 with duration ranging from 45 to 90 minutes each.

Table 5.1 Characteristics of firms interviewed

Case	Type	Firm size	Ownership	Position in supply chain and market segment
1	Fabric manufacture	Micro-small	Private and domestic investment	Fabric supplier for domestic market
2	Fabric manufacture	Small-medium (200 workers)	Private and domestic investment	Fabric supplier for both domestic market (mainly) and foreign market
3	Fabric manufacture	Large Total capacity: 10 million meters per year (dyeing: 100 tons per month)	State investment; Belonging to the Vietnam National Textile and Garment Group (Vinatex)	*At present:* fabric supplier for domestic market *In the future:* fabric supplier for foreign market
4	Fiber manufacture	Large Total capacity: 2500-3000 tons per month	Private and foreign (Korea) investment	Fiber supplier in transnational footwear apparel supply chain
5	Chemical supplier	Large	Private and foreign (Taiwan) investment	Chemical supplier for domestic-market 30%, foreign-market 30%, and exporting 40%

"Tailor-made" and exploratory techniques were adopted in the semi-structured interviews. Tailor-made questions provided space to include the respondent's expertise and experience and firm characteristics. Pre-determined open-ended questions were used to explore the use of NP/NPEOs, the relevant environmental and legal awareness, the possibilities of five responses, and the perceptions of the adaptabilities, risks, benefits, and barriers to the elimination of the surfactants in the textile manufacturing processes. The tentative responses included: using substitute chemicals in place of NP/NPEOs, increasing washing in order to eliminate the residues, monitoring the residues in textile products, waiting for the regulatory enforcement and guidelines from the government, and upgrading wastewater treatment facilities if the limits of NP/NPEOs are integrated into the state regulation. Two categories of technical and organizational adaptabilities were recognized. The main elements of perceived risks included manufacturing processes, product quality, and production cost, whilst the main elements of perceived benefits were profit, market opportunities, and competitiveness. The same question set was asked to the interviewee from the chemical supplier to gain objective perspectives.

To start the interview, the research objective was introduced and the respondent's expertise and firm characteristics were recorded. Second, the respondents were asked about their awareness of the environmental impact, and foreign regulations on eliminating the use of NP/NPEOs. Third, to engage the respondents in the further behavioral questions, they were explained about the environmental situation with regards to NP/NPEOs as well as the adverse effects. Next, the respondents were requested to share their opinions about the possibilities of the aforementioned responses. Attitude was also observed and confirmed by the respondents. What followed were questions to explore how they perceived the adaptabilities, risks, benefits, and barriers that were associated with the responses. The respondents self-expressed their perceptions, where some unclear information (e.g., chemical use, technical and organizational adaptabilities) was re-questioned to confirm. Emphasis in the talk of each respondent was also recorded. Un-structured talks emerged with three respondents about the weaknesses of small-medium-sized textile firms and the market situation of the Vietnamese textile industry.

Information on the market situation and the environmental regulations for the Vietnamese textile industry was collected from a variety of data sources, including published and unpublished documents (e.g., scientific articles, project reports, regulatory documents) and electronic media (e.g., official websites, online news). The following keywords for desktop search were included – but not restricted to – “water (or wastewater) standard”, “tariff rate”, “phenols”, “nonylphenol”, nonylphenol ethoxylates”, “alkylphenols”, “surfactants”, “endocrine disrupting compounds”, “endocrine disruptors”, “restricted substances”, “alternative (or substitute) chemicals” in the domains of “apparel and footwear”, and “textile (and garment) industry”. Another source of information is via un-structured talks with the interviewees, particularly with the expert from the chemical supplier. The questions focus on the types and price of surfactants, as well as quality standards, types, and production cost of fabrics.

5.2.3 Data interpretation and presentation

Cognitive mapping (Axelrod 2015) was employed to represent the relationships between the variables examined in this study. Cognitive maps are useful for modeling complex relationships among variables, hence they have been used to examine decision-making and people’s perceptions in many fields (Özesmi and Özesmi 2004). Özesmi and Özesmi (2004) suggest four ways to construct cognitive maps as follows: from questionnaires, from written texts, from data that shows causal relationships, and through interviews with people who draw the maps directly. In this study, cognitive maps were constructed by the interviewer using written interview data. Since the interviews were based on the pre-determined concepts (or variables), coding was not needed and the process of getting information from the respondents onto paper was straightforward. Causal relationships between the external factors and the internal factors as well as between the elements of the external and the internal factors (as shown in Figure 5.1) were recognized and presented. Cognitive maps were drawn for each firm, and a conceptual model of firm behavior was proposed based on interpreting the relationships in the cognitive maps.

5.3 Results and discussion

5.3.1 Environmental regulatory situation

The lack of knowledge and consequent lack of attention to NP/NPEOs contamination are barriers to environmental and health risk management in Vietnam (Ho and Watanabe 2018). Indeed none of the current regulations specifies NP/NPEOs but regulates the substances under general indicators such as “surfactants” in sewage and textile wastewater, or “phenolic compounds” in drinking water and general industrial wastewater, or both types in surface water quality (MONRE 2008, MOH 2009, 2015, 2011a, b). Since the design of water/wastewater treatment facilities and the monitoring of water/wastewater quality solely rely on these regulations, the control of the environmental contamination of NP/NPEOs is not guaranteed by the existing regulatory system.

On the other hand, global environmental movements that lead to the commitments of mainstream international apparel and footwear corporations to take responsibility of controlling their use of NP/NPEOs potentially benefit the Vietnam environment. After an agreement among 18 mainstream international brands has been obtained, some brands (e.g., Benetton) step up to develop a zero discharge of hazardous chemicals (ZDHC) program, whilst others issue a restricted substances list (RSL) for their stewardship over wet processing suppliers (Benetton group 2017, Levi Strauss & CO. 2015, ZDHC member brands 2014, Puma 2014). In the coming years, foreign regulations, such as the Commission Regulation 2016/26 of the European Union concerning the residual levels of NP/NPEOs in textile articles (EC 2016), are expected to comprehensively reach transnational supply chains with outsourcing factories including those in Vietnam. This implies a diffused regulatory pressure on the Vietnamese textile firms through foreign customers' pressure.

5.3.2 Market situation

Export (foreign) markets have accounted for a large share of the total textile productivity in developing countries such as Vietnam, Bangladesh, Indonesia, and Cambodia (Seyoum 2010). Suppliers in international supply chains must fulfill the requirements of customers. Textile product exports are required to meet strict quality criteria such as absolute color rendering and longevity, UV resistance, waterproofing, electrical resistance (cases 2 and 5, interviews). Most customers do not regulate the types and origins of chemicals, whilst some request ECOTEX certificates for chemicals (case 2, interview). The average production cost for high-quality fabric for light garments (e.g., blouse, shirt) is 12,000 VND/m, equivalent to 0.53 USD/m (case 5, interview).

The domestic market is small and fragile due to competition with the textile products from China and Korea over quality, diversity, and price (cases 1 and 5, interviews). Based on the samples of fabric and colors provided by clients, firms themselves decide chemical use and manufacture processes, as long as the environmental regulations on wastewater, exhausted gas, and solidwaste handling are complied with (case 2, interview). The average production cost for light fabric of medium quality is 9,000 VND/m, equivalent to 0.39 USD/m (case 5, interview).

Most of the supplies of NP/NPEOs (sometimes referred to as detergent agents) are imported (Minh et al. 2014). The chemicals could be easily found in the Vietnamese chemical market under different commercial trademarks such as Tergitol NPn where n ranges from 4 up to 70 (Bang 2015). These chemicals are currently promoted for import in Vietnam with a preferential tariff rate of 0% (MOF 2015) and are sold as mixtures of detergent agents or unmixed types. The dose of unmixed NP/NPEOs in a washing batch may range from 0.5% to 2.5% by weight, depending on the dirtiness of fiber or fabric (case 4, interview). The selling price of detergent agents that contain NP/NPEOs is 33,000 VND/kg (powder), equivalent to 1.45 USD/kg (case 5, interview).

Due to the adverse effects on wildlife and human health, alternative chemicals such as alcohol ethoxylates, octylphenols and their ethoxylates are suggested for use in place of NP/NPEOs (RPA 1999). The performance effectiveness of the alternative chemicals is of concern. According to the Swedish EPA (1998), the alternatives may have lower performance. Nevertheless, the interviewee in case 5 revealed that the substitute chemicals have comparable effectiveness with the traditional detergent agents, and are applicable in current manufacturing processes at equal or higher doses. The selling price of the alternative chemicals ranges from 55,000 to 65,000 VND/kg, equivalent to 2.42 – 2.86 USD/kg (case 5, interview), which is 1.7 – 2.0 times higher than that of the traditional detergent agents.

An estimate by the European Committee of Surface Agents and Organic Intermediates (CESIO) reveals that a complete substitution of NP/NPEOs in the textile and leather sectors would induce 5% of the total costs (RPA 1999). In the wet processes of the cotton, synthetic, and blended fabric manufacturing industry in Vietnam, average water consumption is approximately 168 liters per kilogram of fabric, of which washing water accounts for 75% (Loan 2011). Given that NP/NPEOs and the alternative chemicals are used at the same doses of 0.5% by weight, the consumption of surfactants would be 63 grams per kilogram of fabric, equivalent to 9.5 grams per meter of fabric[4]. Consequently, to substitute NP/NPEOs with alternative chemicals would add 256.5 VND per meter of fabric (equivalent to 0.01 USD/m) into the production cost, increasing it by 1.9% and 2.5% for high and medium quality fabrics, respectively. Comparing to the CESIO's estimate, this is lower but still exerts a pressure on textile firms.

5.3.3 Firm tentative behavior and the determinants

Attitude and tentative behavior

The interview results revealed that none of the firms was aware of the environmental situations regarding NP/NPEOs contamination as well as relevant foreign regulations (e.g., the Commission Regulation). The interviewees showed their attitude and tentative response after the content of the Commission Regulation was introduced. Regulation and market were found to be important drivers of the firm's decision-making regarding the chemical issue, and the impact of regulatory pressure on market pressure was also recognized. Two inverse attitudes and behavioral trends regarding chemical controls were identified.

One trend was an inevitable consequence of using alternative chemicals in place of NP/NPEOs to comply with foreign regulations and to fulfill customer request. This was the only choice for the firm in case 4. The interviewee was concerned about the transnational environmental change under the Commission Regulation force because the firm's market was solely foreign markets. He showed interest when he asked for the hard copy of the Commission Regulation to report to his

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[4] The estimation was based on the reference weight of the light fabric of 135.62 grams per square meter (FabricUK 2015) and the most common widths of a bolt of 112 – 114 cm (Debbie 2017).

manager. To some extent, this implied a positive attitude towards the execution of the foreign regulation.

The other trend was a dependence on the state regulatory enforcement. This was the choice of the firms in cases 1 and 3. They showed no concern about foreign regulations because they were currently suppliers for the domestic market. Meanwhile, there has been no legal and market pressure regarding the restriction of NP/NPEOs on domestic-market-oriented firms.

The firm that took part in two market segments (case 2) revealed both trends. This firm proposed a flexible response to foreign client's request including substituting NP/NPEOs with alternative chemicals for cotton fabric and eliminating the residual level by enhanced washing and reducing the chemical dose for polyester fabric. For domestic market, actions should rely on the state regulation and guidelines as well as customer's request.

In addition, all firms revealed that they were not able to monitor chemical residues due to a lack of facilities. Upgrading wastewater treatment facilities would be considered if NP/NPEOs were integrated in the national standard for wastewater discharge whilst firms continued using the chemicals. Tentative behavior of the firms and the interaction between the determinants are represented in cognitive maps (see Figures 5.2 – 5.5).

Perceived adaptabilities

To initiate the characterization of the dynamics of firm behavior, perceived technical and organizational adaptabilities are described. All cases revealed a technical adaptability since NP/NPEOs were regarded as auxiliary chemicals. Two large-sized firms (cases 3 and 4) emphasized the important role of chemical suppliers regarding the substitution of NP/NPEOs. Accordingly, chemical suppliers would provide catalogue and guidelines about the dose and the manufacturing conditions for alternative chemicals. Substitute chemicals would be tested with fabrics at lab-scale and pilot scale before mass production. Whereas, the firm in case 2 attempted to explore more options for the restriction of NP/NPEOs residues. Using alternative chemicals was considered the best option for cotton since this type of fabric was vulnerable to color bleeding. Since polyester fiber was better in terms of purity, reducing the dosage of NP/NPEOs or enhanced washing in order to eliminate the residues could be an alternative.

Regarding organizational adaptability, the firms in cases 3 and 4 supposed that a minor adjustment would be needed when substituting NP/NPEOs, such as a change in chemical supplier. The firm in case 2 showed concern about incidents that might occur when new chemicals were used in place of conventional chemicals. One way of adaptation was to issue revised procedures after testing with substitute chemicals and adjusting the manufacturing process in order to avoid incidents in mass production. The firm also showed a confidence that technical staffs were able to investigate alternative chemicals of applicable specifications at reasonable quality and price.

Perceived risks

The interview results revealed low perceived risks of product quality deterioration and interference with the manufacturing processes. This was because the firms were able to identify their solutions for both technical and organizational adaptations as previously described. On the other hand, they believed that the alternative chemicals should advance the conventional chemicals.

All firms perceived a risk of increased production cost regarding alternative chemicals. The firm in case 2 showed concern about the cost and perceived this as a risk to its profits. It was private-owned and lacked buffering capacity, therefore a decline in business performance would be unexpected. However, the firm supposed that the cost of alternative chemicals would fall off when the chemicals were widely applied and that the price of alternative chemicals depended on the time and the quantity of orders. This view underpinned a positive attitude to comply with foreign clients' request in case 2. Nevertheless, those firms in cases 1 and 3 perceived that using alternative chemicals was impractical for domestic market because this would yield an increased production cost, and thus lower their competitiveness. Since the action of chemical substitution might be considered if clients requested, the firm in case 1 also perceived a risk of manufacturing delays due to the need for testing alternative chemicals.

Perceived benefits

The motivation behind the tentative behavior of cases 2 and 4 was that they perceived potential benefits from foreign markets. Compliance with foreign regulations could enhance the firms' competitiveness with other suppliers; therefore, they could at least maintain their positions in the current supply chains. The firm in case 2 even viewed the Commission Regulation as a challenge for all firms owning positions in transnational supply chains; thus there would be an opportunity for enlarging the foreign market segment if the firm could overcome the challenge. Having experience in the textile industry, the interviewee in case 5 revealed that the stability of firm's business performance would be more guaranteed if they could participate in transnational supply chains, which were considered as high market segments. In his view, high market segments could accept an alleviation in selling price due to an increased production cost for environmentally friendly garment items, which would sustain fabric suppliers. Additionally, to comply with international standards (e.g., REACH, BLUESIGN, ISO 9001/14000, OSAS 18000), firms would gain trust from clients and in turn more comfort in internal authority.

The firms in cases 1 and 3 perceived no benefit from the domestic market due to increased production cost and high competition. They also shared the belief that the firms might take the implementation of foreign regulations as an opportunity to reach new markets in the future. Firm's business performance was supposed to decline at the beginning due to the increased cost associated with alternative chemicals, but it should be able to revive when alternative chemicals are popularly used.

Perceived barriers

The firms in cases 1 and 2 perceived a lack of human resources necessary to keep pace with environmental changes. It was because information on chemical use was provided by chemical suppliers and the firm leaders. Whilst state-capital firms could be supported by the Vietnam Textile and Apparel Association (VITAS) in terms of information and international trade, private firms could hardly access that service. No facility for testing the residues of NP/NPEOs was perceived as another barrier to monitoring activity. The firm in case 1 showed preference for invariable performance, which was perceived as a barrier to the likeliness of adaptation to changes. Notwithstanding, the firm in case 3 seemed to depend on its strategic plan regarding target markets that would drive the corresponding manufacturing process and chemical use.

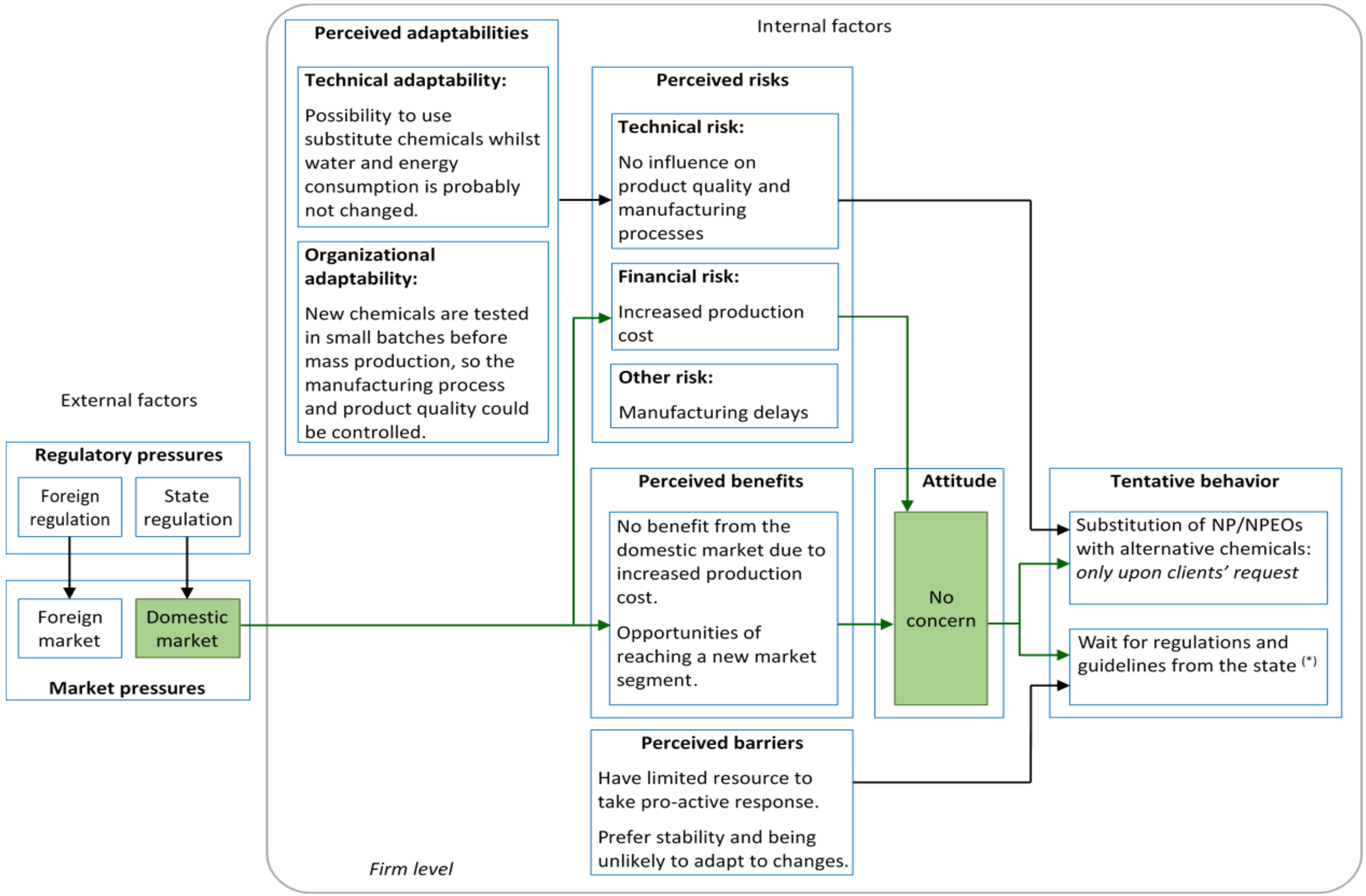

Figure 5.2 Cognitive map of the micro-small-sized firm's behavior (case 1)

Notes: (*) This behavior includes upgrading wastewater treatment facilities if needed when NP/NPEOs are integrated into the state regulations.

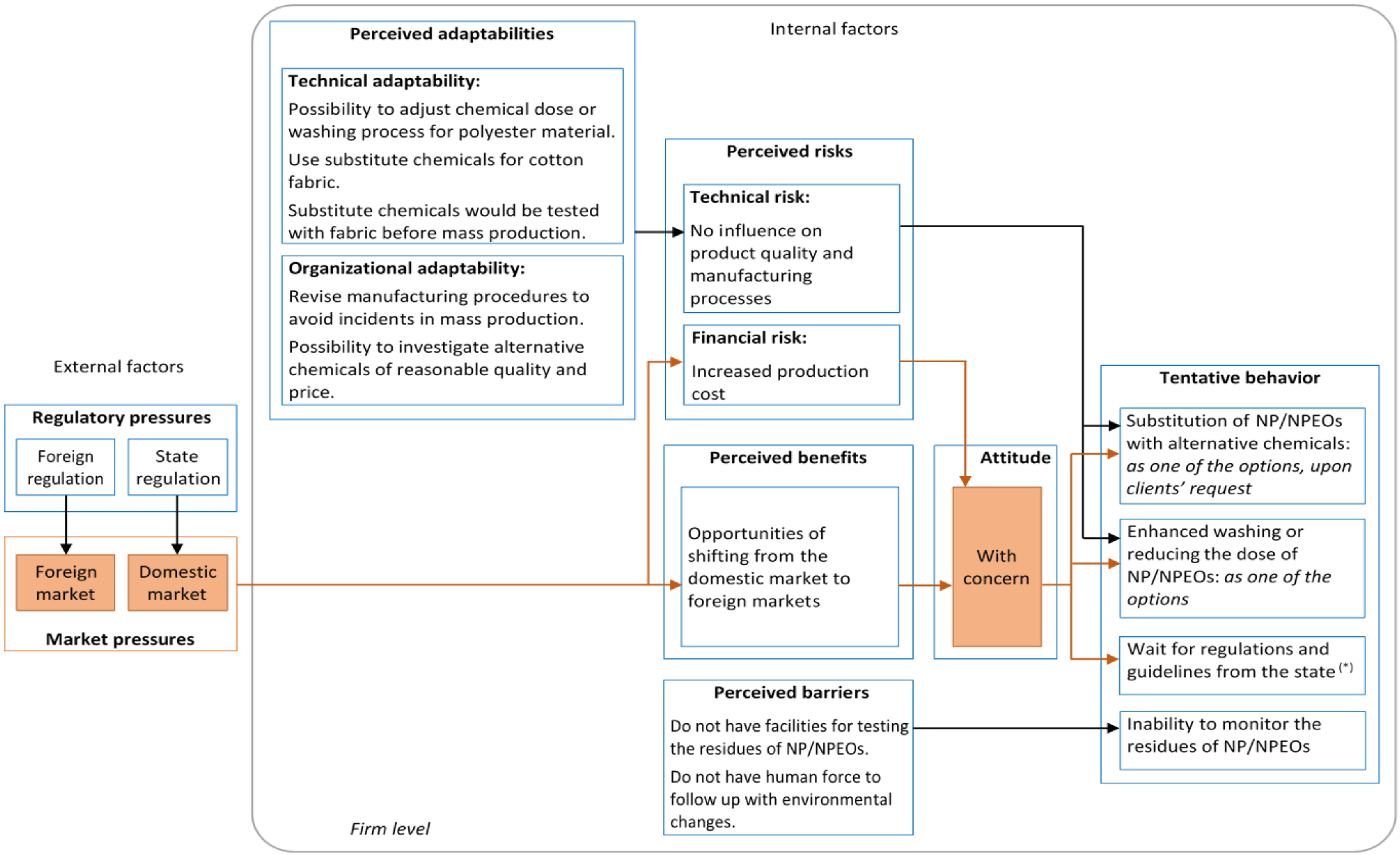

Figure 5.3 Cognitive map of the small-medium-sized firm's behavior (case 2)

Notes: (*) This behavior includes upgrading wastewater treatment facilities if needed when NP/NPEOs are integrated into the state regulations.

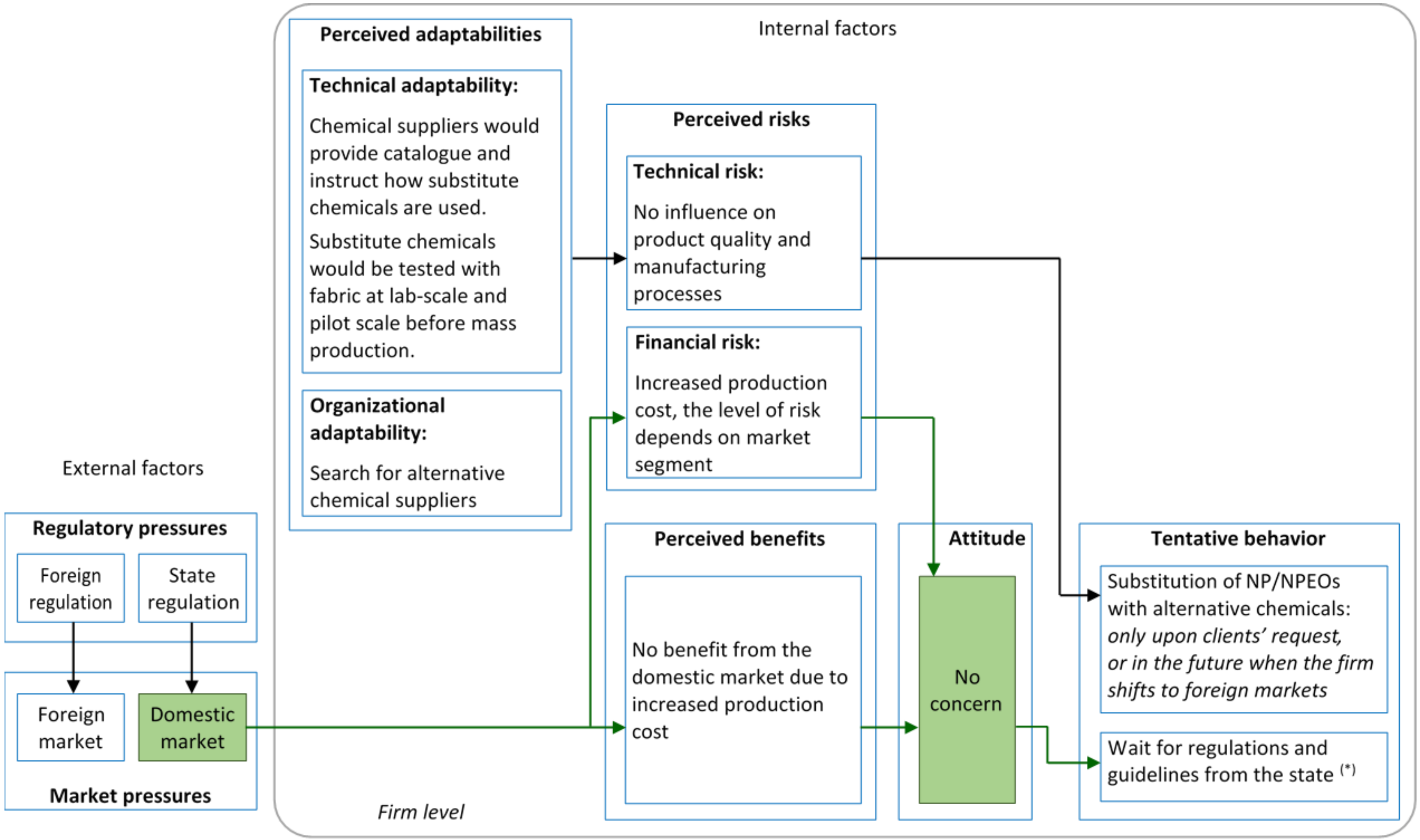

Figure 5.4 Cognitive map of the large-sized state-capital firm's behavior (case 3)

Notes: (*) This behavior includes upgrading wastewater treatment facilities if needed when NP/NPEOs are integrated into the state regulations.

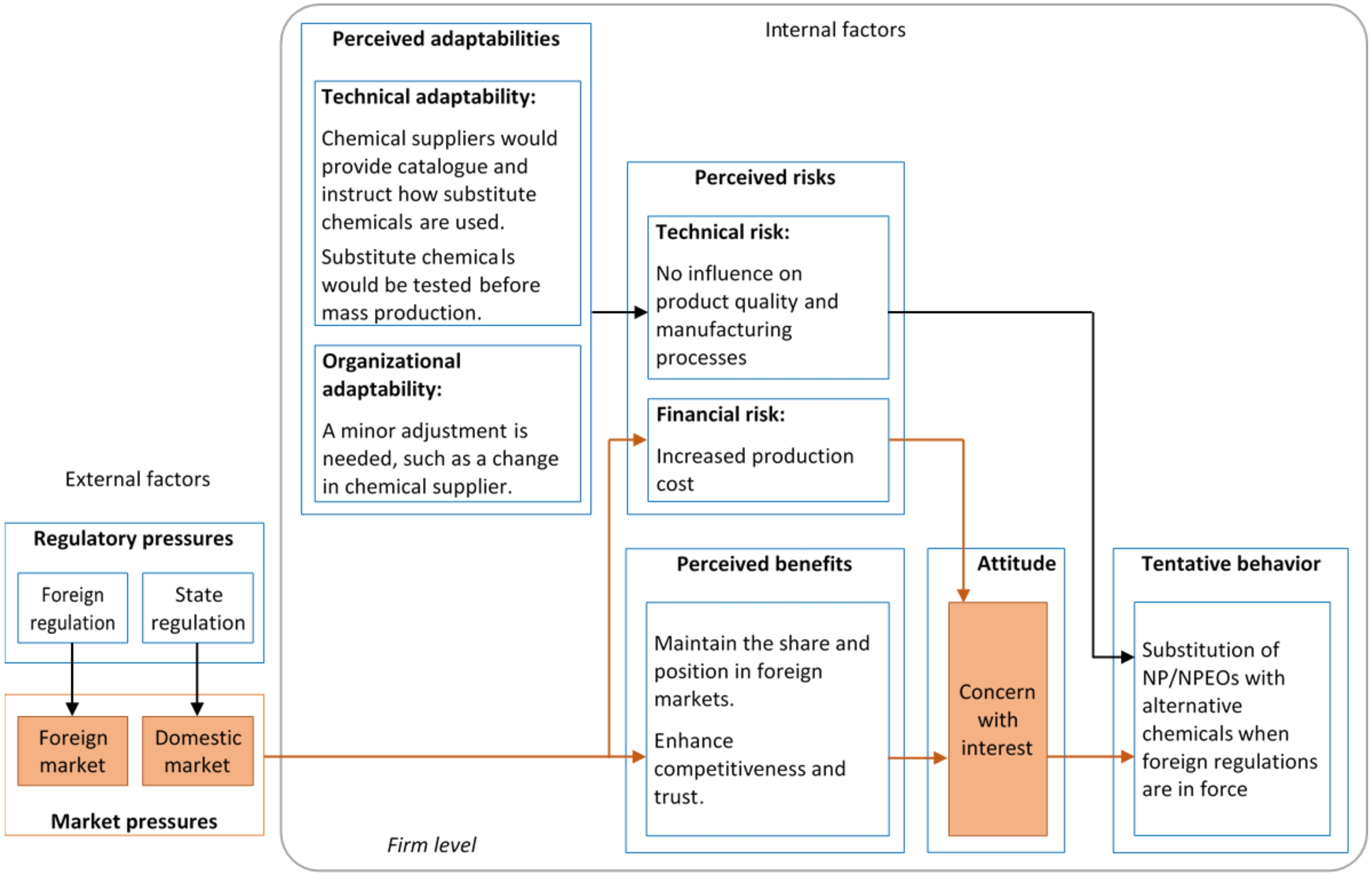

Figure 5.5 Cognitive map of the large-sized private-capital firm's behavior (case 4)

5.3.4 The moderation role of market segment

The interviews result in two inverse attitudes towards NP/NPEOs restriction: with concern (cases 2 and 4) and without concern (cases 1 and 3), the explanation of which may relate to the perceived risks and benefits and market strategy (summarized in Table 5.2). It could be seen that the positive benefit perception of the foreign markets has driven the concerning (positive) attitude of the respondents in cases 2 and 4, while the respondents in cases 1 and 3 perceive no benefit from the domestic market and in turn a negative attitude without concern. Noticeably, the positive benefits of cases 1 and 3 are perceived conditionally only if those firms decide to switch to or take part in foreign markets. In addition, although increased production cost is recognized as a financial risk by all respondents, it is particularly regarded as a responsibility for the infeasibility of chemical substitution among the domestic market-oriented firms. The interpretation suggests that market strategy may play a moderating[5] role in the relationship between perceived risks and benefits and attitudes towards the restriction of NP/NPEOs.

Table 5.2 Influence on firm's attitude

Factor	Case 1	Case 2	Case 3	Case 4
Perceived financial risk	--	-	--	-
Perceived benefits	-/+	+	-/+	+
Overall attitude towards the restriction of NP/NPEOs	No concern (-)	With concern (+)	No concern (-)	Concern with interest (++)
Market strategy	Domestic	Domestic & foreign	Domestic	Foreign

Notes: "-" and "+" indicate negative and positive impacts on attitude, respectively; "--" and "++" indicate higher levels of perceived financial risk and concern, respectively.

5.3.5 Indirect influence on firm's response

The qualitative study allows the interviewees to express their perceptions with attitudes and explanations that really make sense to their response on tentative behavior. The causal-effect relationships between the adaptabilities and risk elements have been identified from the responses of the interviewees. Accordingly, the positive perception of technical and organizational adaptabilities has been found to explain the perception of low risks regarding product quality and manufacturing processes and a risk of delayed manufacture due to testing alternative chemicals. Although the relationship between the perceived risk of adopting environmentally friendly technology and overcompliance behavior has been identified (Wu 2009), the indirect influence of perceived adaptabilities through perceived technical risks on firm tentative behavior has not been mentioned in the existing literature. As aforementioned, since this finding was based on a

[5] A factor is a moderator when it exerts a strengthening or weakening impact on the relationship of one variable with another (Zhu et al. 2014).

qualitative research, more weight and quantitative methods would be encouraged for further studies for complementary purpose (Hussein 2015).

5.3.6 A modified conceptual model of firm behavior and theoretical implications

Elaborating on the literature in the field summarized in Figure 5.1, and the interpretation of the findings, a modified conceptual model of firm behavior is proposed in Figure 5.6. The model comprises two external factors as regulatory enforcement and market pressure influencing firm's attitude and response patterns through the perception of financial risk and benefits. Market strategy is supposed to impact the extent to which perceived financial risk and benefits influence attitude and, in turn, its tentative response. Perceived technical and organizational adaptabilities explain perceived technical risks that drive the (un-)likeliness of taking certain responses, while perceived barriers directly explain why certain responses are not feasible. In this model, perceived technical risk and barriers have not been found to explain attitude, probably because they play smaller roles in the direction of decision. The findings are in line with Shipley (1981) who has found that manufacturing firms perceive profit as being very important compared with other goals. Additionally, substitution of NP/NPEOs is perceived as a minor adjustment in the textile manufacturing processes in this study.

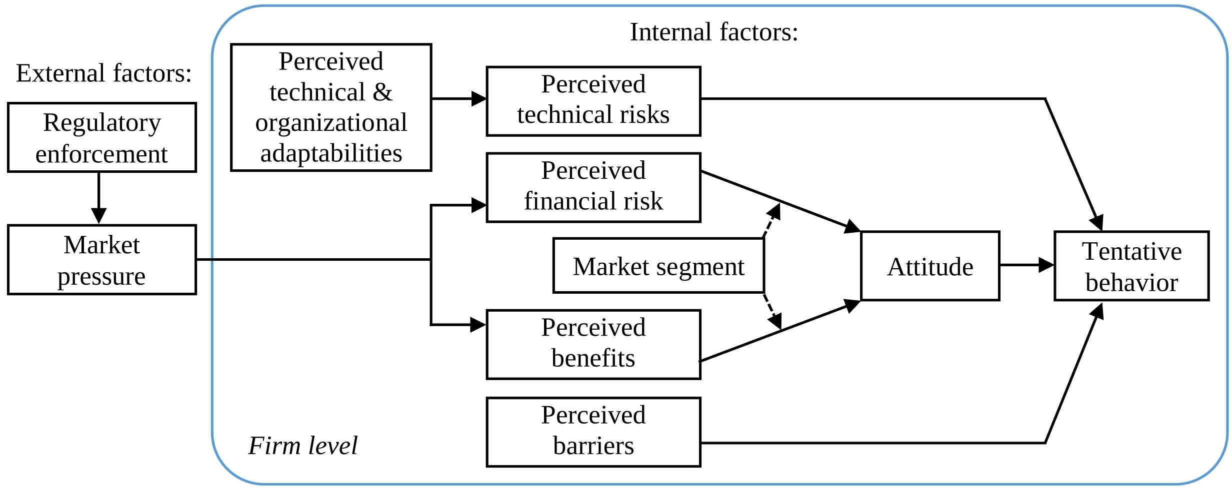

Figure 5.6 A modified conceptual model of firm behavior

The proposed model could be useful for studying firm behavior in developing countries in the early stage of globalization when the gaps in production technology and in environmental protection are still wide. The pulp and paper, plastic, and agricultural processing are the industries of interest. The model could also be useful for feasibility studies and effectiveness assessment of environmental policy at micro (firm/ industry/ community/ household) level. An example is the study of the feasibility of tightening sewage/ industrial discharge/ emission quality standard, where the policy of enhancing environmental protection may influence the existing treatment facilities and in turn firm environmental performance. Another example is the study of the effectiveness of a solid waste separation program, in which success may depend on how people perceive adaptabilities, benefits, risks, and barriers. In these cases, another moderator (e.g., education) instead of market strategy is suggested.

5.3.7 Policy implications

The findings have revealed that information shortage is one of the barriers for private firms. Therefore, a social policy to promote the establishment of non-governmental textile and garment industrial/trade associations that help strengthen firm's capacity for information is needed. It is believed that information is the most salient resource of firms (Wernerfelt 1984), where such associations could offer support in the form of useful information on regulatory and market changes, exporting requirements, as well as the diffusion of new regulations, technological innovations, and chemicals (case 2, interview). The policy should aim at firms participating in transnational supply chains so that they can on the one hand comply with foreign regulations, and on the other hand reduce negative impacts on the environment.

The study has also revealed that firms may take alternative actions such as reduction of chemical dosages or increase in washing. If this is the case, the discharge of NP/NPEOs remains as a burden in the Vietnamese environment. Therefore, further studies should investigate the available wastewater treatment facilities in the textile industry in order to assess the distribution and removal of NP/NPEOs, with a focus on those plants discharging effluents directly into the environment. Such studies would assist in prioritizing actions for balancing between economic growth and environmental protection.

5.4 Conclusion

The study revealed diverse responses to chemical elimination where perceived technical risk, financial risk, benefits, and barriers played different roles in explaining a certain action. The attitude towards chemical restriction was shaped by the trade-off between perceptions of financial risk and benefits and was moderated by market strategy. The insight of the textile firm behavior is beneficial for regulatory agencies in designing environmental risk management strategies as well as reinforcing the textile exporting capacity of Vietnam. In addition, a conceptual model explaining diverse responses for firms was initiated, which could be useful for the studying of firm behavior in developing countries in the early stage of globalization and the feasibility and effectiveness of environmental policy at micro level.

To the best of our knowledge, this is the first study to examine the mechanism of diverse behaviors among textile firms towards the restriction of NP/NPEOs, although studies on single behaviors and other aspects of the chemicals are plentiful. The study employed triangulation of data sources which help improve the validity and usefulness of the information.

The small number of interviewees is the limitation of the study. It is because most of the firms in Vietnam are hesitant to talk about sensitive topics such as waste treatment and chemical use. Future studies with participation from a larger number of firms will enable generalization of results.

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CHAPTER 6

DISTRIBUTION AND REMOVAL OF NONYLPHENOL ETHOXYLATES AND NONYLPHENOL FROM TEXTILE WASTEWATER - A COMPARISON OF A COTTON AND A SYNTHETIC FIBER FACTORY IN VIETNAM

Summary

The textile industry is a significant source of nonyphenol and their ethoxylates, which are suggested as responsible for endocrine disruption in wildlife and humans. This study is a comparison of two conventional wastewater treatment processes in a cotton and a synthetic fiber factory in Vietnam, with regard to the distribution and removal of nonyphenol ethoxylates and nonyphenol across each process. Diverse trends in the distribution of nonyphenol ethoxylates in wastewater that distinguished in materials could be revealed. Primary coagulation might not perfectly facilitate nitrification in the secondary activated sludge process regarding pH. Nevertheless, satisfactory removals were achieved during coagulation as well as activated sludge processes in both systems. The roles of long hydraulic retention times (as 21 and 16 hours), low organic loadings (as 0.1 and 0.2 gCOD/gMLVSS.day), extended solids retention times (as 61 and 66 days), and mixed liquor suspended solids of greater than 2,000 mg/L have been demonstrated. The findings provide evidence and a better understanding of nonyphenol ethoxylate and nonyphenol removal efficacy as well as influencing factors in Vietnamese textile wastewater treatment. The results are beneficial for the textile industry in Vietnam regarding investment decisions for wastewater treatment.

6.1 Introduction

Vietnam is one of the top ten textile exporters in the world. From 2004 to 2014, the textile industry achieved a compound annual growth rate (CAGR) of about 19% per year, increasing its contribution to Vietnam's GDP by 5% to 15% (Nguyet 2015). About 62% of factories are established in southern Vietnam (Tot 2014). Raw materials in the textile industry include natural fibers (cotton, wool, silk), synthetic fibers (nylon, polyester, viscose), and a blend of natural and synthetic materials (Loan 2011, Zhang 2014). The textile industry is one of the most polluting industries releasing highly toxic and persistent chemicals into the environment, especially the watercourse (Hasanbeigi and Price 2015, Loan 2011). It is suggested that nonylphenol ethoxylates (NPEOs) are still widely used in the textile industry (Antal et al. 2016) as detergents and auxiliaries in wool scouring, hydrogen peroxide bleaching (Loos et al. 2007), washing, dyeing, and printing (Cobbing et al. 2013, Munn 2011). Studies by Brigden et al. (Brigden et al. 2012b, Brigden et al. 2013a, Brigden et al. 2012a) have demonstrated the worldwide presence of NPEOs in the majority of textile products regarding all materials and across most of the countries of manufacture and consumption. The authors also suggest that lower traces of NPEOs in textile products may imply a higher discharge of NPEOs into the watercourse during production.

NPEOs are indirectly responsible for endocrine disruption among wildlife and human beings via their metabolites, especially nonylphenol (NP) (EC 2002, USEPA 1996). NP, on the one hand, shows to be persistent at different levels in the environment (Yoshimura 1986, Manzano et al. 1999, Ejlertsson et al. 1999). On the other hand, NP has a high affinity for solids such as sediment, sewage sludge, and soil amended with sewage sludge (Ahel et al. 1994a, Ahel et al. 1994b), as well as for lipids (Ademollo et al. 2008), hence it has been shown to accumulate in organisms (Ahel et al. 1993, Ying 2006). Past research has demonstrated the presence of NP in all trophic levels such as plankton, benthic invertebrates, fish, birds, and mammals (Casatta et al. 2015, Casatta et al. 2016, Diehl et al. 2012, Gu et al. 2016, Hu et al. 2005, Isobe et al. 2007, Korsman et al. 2015), via a complex food web (Kidd et al. 2012). NP has been detected in foodstuffs (Chen et al. 2010, Gu et al. 2016, Guenther et al. 2002, Gyllenhammar et al. 2012, Lu et al. 2007), drinking water (Shao et al. 2005), human adipose tissue , urine , maternal blood plasma and amniotic fluid , blood serum , and breast milk (Ademollo et al. 2008, Azzouz et al. 2016, Chen et al. 2010). The presence of NP in pregnant women's decidua and early embryos along with maternal transfers has also been observed (Chen et al. 2016).

Adverse effects of NP on reproductive, immune, and central nervous systems have been discovered in fish, rats, birds, and humans with possible abnormalities in embryos and offspring (Cosnefroy et al. 2009, Ghisari and Bonefeld-Jorgensen 2005, Jie et al. 2010, Jobling et al. 1996, Kim et al. 2006, Mao et al. 2010, Nakazawa and Ohno 2001, Pedersen et al. 1999, Razia et al. 2006, Soto et al. 1991, Vosges et al. 2012, Couderc et al. 2014, WHO-UNEP 2013). Recent studies on carcinogenesis have reflected the relation of exposure to NP to the possibilities of breast cancer in women (Wu et al. 2008) and prostate cancer in men (Forte et al. 2016, Kim et al. 2016). A study of Lepretti et al. revealed negative impacts on human intestinal homeostasis and functionality. The mechanisms of action of NP are related to xenoestrogens (Pillon et al. 2005, Mnif et al. 2007, Kuiper et al. 1998), antiestrogens (Preuss et al. 2010), and disruption of thyroid function (Hofmann et al. 2009), which occur on nuclear (genomic) (Wu et al. 2008), extranuclear (non-genomic) (Thomas and Dong 2006), and cross-talk between genomic and non-genomic pathways (Li et al. 2006, Bulayeva and Watson 2004).

Concerning the negative impacts on the ecosystem and human beings, NP/NPEOs have been added to the list of chemicals for priority action since 1998 by the OSPAR Commission (OSPAR 1998). From 2000, NP has been classified as a priority hazardous substance under the Directive 2000/60/EC of the European Parliament and the Council. Marketing and use restrictions of NP/NPEOs have also been put into place under the Directive 2003/53/EC, and the concentration of NP in surface waters has been regulated in the Directive 2008/105/EC. Accordingly, the annual average level of NP should not exceed 0.3μg/L. Nevertheless, NP and NPEOs are allowable for use today in many Asian countries including Vietnam.

In Vietnam, NP has been detected at extremely high levels in urban watercourses such as in the cities of Ha Noi and Ho Chi Minh, in the range of 0.02 – 9.7 μg/L (mean 3.0 μg/L) and 2.0 – 20.0 μg/L (mean 9.7 μg/L), respectively (Hanh et al. 2014). Regarding environmental risk, it is suggested that NP may cause ecological effects due to its high risk quotient (MEC/PNEC[6] = 128) (Chau et al. 2015). Recent investigations in textile manufactures in developing countries such as Thailand, China, Mexico and Indonesia by Greenpeace have revealed that NP/NPEOs are among the most commonly detected hazardous chemicals in the effluent of wastewater treatment plants (WWTPs) (Cobbing et al. 2013). NP and NPEOs with one and two ethoxylate group(s) have been identified as the most dominated alkylphenols in the effluent of PT Gistex, one of the biggest textile factories in Indonesia (Brigden et al. 2013b). Brigden et al. reported a level up to 14 μg/L of NP at the effluent of Youngor Textile Complex, Yangtze River Delta, China. These concentrations of NP and NPEOs do not represent Vietnam but they illustrate the current problematic condition of textile production in developing countries in Asia (Cobbing et al. 2013).

Every textile manufacturer in Vietnam utilizes WWTPs. However, their designs are for addressing only macro-pollutant issues with a major application of conventional processes (VEA 2011). It is suggested that conventional processes are inferior to effectively remove NPEOs and NP in wastewater (Brigden et al. 2013b, Zhou et al. 2009). Therefore, the effective elimination of endocrine disrupting compounds is a great challenge for the Vietnamese textile industry. More investigations on the existing textile WWTPs in Vietnam regarding the removal of nonylphenol are needed. On this basis, the objectives of this study are:

- To investigate the distribution and removal of NPEOs and NP across two typical textile wastewater treatment processes.
- To assess the influence of hydraulic retention time (HRT), nitrifying conditions, solids retention time (SRT), mixed liquor suspended solids (MLSS), and water temperature on the NPEO and NP removal.

6.2 Treatment processes and factors influencing nonylphenol and nonylphenol ethoxylate removal

In aqueous phase, NPEOs undergo biotransformation as a predominant process under aerobic condition (Ahel et al. 1994c, Mann and Boddy 2000, Manzano et al. 1999, Potter et al. 1999, Scott and Jones 2000) and partly under anaerobic condition (Ahel et al. 1994a, Charles et al. 1996). Consequently, metabolites such as nonylphenol mono- and di- ethoxylate ($NP_{1-2}EO$), nonylphenol mono- and di- ethoxy carboxylate ($NP_{1-2}EC$), and the final refractory product NP are formed (Ahel et al. 1994a, Ahel et al. 1994b, Di Corcia et al. 1994, Field and Reed 1996, Staples et al. 1999). They have great hydrophobicity except for $NP_{1-2}EC$ and less biodegradability. Due to their

[6] MEC: Measured Environmental Concentration; PNEC: Predicted No-Effect Concentration; MEC/PNEC ratio of greater than 1 indicates a potential hazard or risk.

lipophilicity, NP and $NP_{1-2}EO$ can be eliminated via biotransformation and sorption (Ahel et al. 1994a, Cirja et al. 2008, Fauser et al. 2001). Although a variety of wastewater treatment processes has been studied for their ability of removing nonylphenolic compounds from wastewater, it seems that conventional processes still dominate in practice. Primary coagulation/flocculation followed by a secondary anaerobic/anoxic/aerobic activated sludge has shown to be popularly applied in textile industry (Grau 1991), particularly in Vietnam (Loan 2011, VEA 2011). The removal of nonylphenolic compounds depends on the nature of the processes (Zhou et al. 2009) and factors such as the population served by the sewage system (Castiglioni et al. 2006), wastewater compositions (Cirja et al. 2008, Pothitou and Voutsa 2008), hydraulic retention time (HRT) (Johnson et al. 2005), biomass concentration (expressed as MLSS), solids retention time (SRT), pH and temperature (Cirja et al. 2008). Table 6.1 demonstrates the removal efficacies of NPEOs and NP by some commonly used wastewater treatment processes.

6.2.1 Removal by coagulation process

The main implication of coagulation is to remove total suspended solids and colloids in wastewater (Ejlertsson et al. 1999). In textile wastewater treatment, taking the advantage of particle removal, coagulation also functions for elimination of dye agents that are microbial inhibitory and attribute to the color of wastewater (Loan 2011). Regarding nonylphenolic compounds, coagulation process has shown its capacity of removing NP, NP_1EO and NP_2EO (Ahel et al. 1994a), which are considered hydrophobic with high partition coefficients (log K_{ow}) of 4.48, 4.17, and 4.21, respectively (Ahel and Giger 1993). High removals of over 90% of NP (Vogelsang et al. 2006), and over 90% of $NP_{1-2}EO$ (Pothitou and Voutsa 2008) were achieved during this process. Coagulation and flocculation efficiencies are dependent on wastewater compositions, coagulants, and process conditions such as pH and mixing (Metcalf and Eddy 2004).

6.2.2 Removal by anaerobic processes

Under anaerobic condition, a complete de-ethoxylation of NPEOs with the formation of NP could be obtained (Ahel et al. 1994a). Results of Zhou et al. (2015) demonstrated up to 200% of NP that was formed from the conversion of NPEOs in anaerobic units, suggesting the elimination of NP by this process was negligible. In contrast, anaerobic biological process was shown to contribute a great deal in NP removal, at 82.5%, over the total removal efficiency of the whole biological system, of 89% (Tan et al. 2007). Similarly, a majority of NP being removed in the anaerobic bioreactor, a rate of 42% over the total efficacy of 67% through a combined anaerobic-activated sludge process was reported (Zhou et al. 2009). Diverse potencies have been documented probably because the biotransformation of NP in anaerobic condition may strictly rely on the availability of nitrate in wastewater as an electron acceptor (Luppi et al. 2007). Indeed, the study of Wang et al. (2013) revealed that NP was mainly removed via sorption on sludge, and no biodegradation was observed during an anaerobic-without-nitrate condition, suggesting the dominant role of nitrate in NP removal via anaerobic processes.

6.2.3 Removal by anoxic processes

Literature on the elimination of NPEOs and NP in anoxic condition is scarce. However, Lee Ferguson and Brownawell (2003) suggested that only anoxic condition was able to complete de-ethoxylation of NPEOs. However, anoxic degradation of NPEOs might not last long since it required a combination of an electron acceptor and a specialized microbial consortium (Luppi et al. 2007) or denitrifying conditions (Lu et al. 2008). Lu et al. (2008) suggested a more efficient removal of NPEOs by denitrifying activated sludge than by anaerobic activated sludge. Regarding NP, Zhou et al. (2012) observed an approximately 82% reduction in NP within 8 hours. Whereas, Wang et al. (2013) reported a removal of 89% after 120 hours. The authors demonstrated a rise in removal efficacy at an increased nitrate concentration, where the most suitable COD/nitrate ratio was of 15:1. It may imply that the degradation of NP is also facilitated by denitrification.

6.2.4 Removal by aerobic activated processes

Removal efficiency of NPEOs and NP in wastewater by aerobic ASPs has been demonstrated by numerous scholars (Table 6.1). Tan et al. (2007) revealed that conventional activated sludge systems could be particularly effective and had a NP removal potential of 85% to 99% in the influents comprising municipal, industrial and hospital wastewater in mixture. Zhou et al. (2015) affirmed the major role of ASPs which contributed 93 – 94% over total NP removals of three denim WWTPs. Though biotransformation dominate in anaerobic and anoxic processes, it appears in aerobic ASPs that the removal of nonylphenolic compounds takes place via biodegradation and wastage of excessing sludge (Nie et al. 2014, Zhou et al. 2012). Accordingly, a major removal of long-chain NPEOs (with logKow less than 2.5) is via biotransformation (Langford et al. 2005a), whereas removal of NP and short-chain NPEOs (with logKow greater than 4) is more likely via adsorption onto biomass (Gao et al. 2014, Langford et al. 2005b). Factors such as HRT, nitrification, organic loading, SRT, biomass concentration (expressed as MLSS), temperature, and pH have been suggested significant for governing those mechanisms (Cirja et al. 2008, Langford et al. 2007, Loyo-Rosales et al. 2007b, Scruggs et al. 2004).

Table 6.1 Removal efficiency of nonylphenol and nonylphenol ethoxylates

Substance	Anaerobic process[a]/ Coagulation process[b]	Activated sludge process	Conditions	Source
NP	82.5%[a]	6.5%	Anaerobic followed by aerobic activated sludge	Tan et al. (2007)
NP	42%[a]	25%	Anaerobic activated sludge followed by oxidation ditch SRT~16-17 days	Zhou et al. (2009)
NP	Negligible	93 – 94%	Anaerobic followed by aerobic activated sludge* HRT*~7.8-9.8 hours SRT*~10.5 – 11.9 days MLSS*~3,500-4,000 mg/L	Zhou et al. (2015)
NP		~99%	Activated sludge Long SRT~25 days	Ivashechkin et al. (2004)
NP		97 – 99%	Activated sludge with nitrifying Ammonia removal~62 – >90% HRT~10 hours SRT~3-20 days MLSS~500-3,000 mg/L	Stasinakis et al. (2010)
NP $NP_{1-2}EO$		80% >90%	Activated sludge Temperature~20-30°C MLSS~1,676 mg/L HTR~64 hours Relatively low loadings	Pothitou and Voutsa (2008)
NP $NP_{1-3}EO$ $NP_{4-12}EO$		70% 93% 93%	Activated sludge Long SRT~27 days HRT~16 hours	Petrie et al. (2014)
NP NP_1EO NP_2EO NP_6EO		37% -3% (produced) -5% (produced) 78%	Activated sludge High loading/ non-nitrifying	Birkett and Lester (2003)
NP NP_1EO NP_2EO NP_6EO		77% 31% 91% 98%	Activated sludge Low loading/ nitrifying	Birkett and Lester (2003)
NP and NPEOs		>80%	Activated sludge Low loading/ nitrifying STR>10 days	Kreuzinger et al. (2004)
NP and $NP_{1-2}EO$		>90%	Activated sludge Low loading/ nitrifying-denitrifying	Clara et al. (2007)
NP	>90%[b]		Coagulation	Vogelsang et al. (2006)
NP NP_1EO NP_2EO	53%[b] 91%[b] 94%[b]		Coagulation	Pothitou and Voutsa (2008)

6.3 Materials and methods

6.3.1 Selection of wastewater treatment processes

Surveys in 120 textile manufactures in Vietnam in 2010 revealed two types of WWTPs (VEA 2011). For the first type, a primary coagulation followed by a sand or powder activated carbon filtration was connected to the central WWTP of an industrial zone. The second type included

WWTPs which effluent wastewater was discharged directly into the watercourse. These WWTPs employed a biological process in addition to physico-chemical processes. An investigation of forty textile WWTPs in southern Vietnam revealed that wastewater treatment processes for cotton or blended fabrics mainly involved the physico-chemical process (coagulation) and the activated sludge process (ASP), whereas those for polyester required both primary and tertiary physico-chemical processes in combination with a secondary biological process. In some cases, ozone oxidation, filtration, or electrochemical processes were used in place of tertiary treatment (CENTEMA 2010). In general, ASP is the most widely used wastewater treatment process in the Vietnamese textile industry (Loan 2011).

Removal of micropollutants is governed by factors such as physico-chemical properties of substances, wastewater compositions and characteristics, and wastewater treatment processes linked with treatment conditions (Luo et al. 2014). With given properties of NPEOs and their metabolites, differences in the nature of wastewater and treatment processes for cotton/ blended fabrics and synthetic materials were of interest.

Our criteria for selecting WWTPs for the investigation were inherited from the outcome of the surveys by VEA (2011) and CENTEMA (2010), taking into account the influent concentrations of NPEOs and NP as well as treatment processes as previously discussed. Since there is no regulated limit for the discharge of NP in Vietnam, it is supposed that WWTPs that directly discharge into waterbodies might pose a threat to the aquatic ecosystem. Therefore, one textile WWTP connecting to the central WWTP of an industrial zone and another connecting to a river were selected. Second, the textile factories should involve a dying – finishing process. This process comprises three main steps as preparing, dyeing or/and printing, and finishing, in all of which surfactants and auxiliaries are used for impurity removal, enhancement of color agent dispersion, abrasion resistance and improvement of the tear strength of fabric (Loan 2011). Lastly, materials of each factory should be typically different.

Consequently, among some factories that met the three aforementioned criteria, permission from two factories to carry out the survey was granted. Hoa Sen factory was established in 2002 and is specialized in the production of cotton fabric. Chyang Sheng factory has been in operation since 1996 and produces synthetic garment products. Both are Taiwan-owned factories. The WWTPs of both factories employ a physico-chemical process followed by a biological process. The WWTP of Hoa Sen factory, namely as F1, adopts a primary coagulation prior to a consecutive ASP and disinfection. For the WWTP of Chyang Sheng factory, namely as F2, a primary coagulation is followed by a secondary ASP and a tertiary coagulation before disinfection.

6.3.2 Field survey and sampling

The survey campaign took place from September 5th to September 7th, 2016, at Hoa Sen and Chyang Sheng. Since this is the first investigation with the primary purpose of demonstrating the presence (with distribution) or absence of nonylphenolic compounds in textile wastewater in

Vietnam, grab sampling could be sufficient (Ort et al. 2010). Sampling was performed as suggested by Pothitou and Voutsa (2008). On each day, grab wastewater samples were collected from the equalization tanks and at the outlets of treatment facilities such as primary clarifiers (following coagulation process), secondary clarifiers (after ASP), and disinfection tank (for F1) or tertiary clarifier (after advanced treatment by coagulation, for F2). The wastewater containing activated sludge was collected directly from the aeration tanks, and the returning sludge sample was collected from the sludge outlets of the secondary clarifiers. Wastewater samples were stored in 1L glass bottles, which had been well rinsed with acetone solution, followed by Mili.Q solution, and finally evaporated by a drying facility. Sludge samples were stored in 1L plastic bottles. All samples were kept in ice containers when they were conveyed to laboratories on the day. the Measurements of temperature, pH, and DO were conducted on site. Sampling positions are shown in Figure 6.1. Treatment processes and technical specifications such as design capacity and operating capacity, facility volume, hydraulic retention time, sludge returning schedule, and chemical use of the two wastewater treatment facilities were also investigated.

The estimations of solids retention time (SRT), returning activated sludge ratio, organic loading, and removal rate were detailed as follows.

Traditional mass-balance approach for the estimation of SRT (day) (Metcalf and Eddy 2004):

$$SRT = \frac{V_A X_A}{(Q - Q_W)X_E + Q_W X_R}$$

Where V_A denotes the volume of the aeration tank (m^3). X_A, X_E, and X_R indicate the concentration of biomass in the aeration tank (g/m^3), the concentration of biomass in the effluent (g/m^3), and the concentration of returning activated sludge (g/m^3), respectively. In our study, the effluent total suspended solids (TSS) was assumed as X_E, and the mixed liquor suspended solids (MLSS) was as X_A. Q and Q_W represent the wastewater flowrate (m^3/day) and the waste activated sludge flowrate (m^3/day), respectively.

Estimation of returning activated sludge ratio (R) (Metcalf and Eddy 2004):

$$R = \frac{1 - (HRT/SRT)}{(X_R/X_A) - 1}$$

Where HRT denotes the hydraulic retention time in the aeration tank (day).

Estimation of organic loading (F/M, g/g.day) (Metcalf and Eddy 2004):

$$F/M = \frac{QS_o}{VX}$$

Where S_o, V, and X indicate influent concentration (g/m^3), volume (m^3), and biomass concentration (g/m^3), respectively. In this case, S_o was the filtered COD in the influent, V was equal to V_A, and X was the mixed liquor volatile suspended solids (MLVSS) in the aeration tank.

Estimation of removal rate (%):

$$\text{Removal rate } = \frac{(S_o - S_e)\times 100}{S_o}$$

Where S_o and S_e represent the influent and the effluent concentrations (g/m^3), respectively.

Nonylphenol ethoxylates are denoted as NPEOs or NP_nEO, where n indicates the numbers of ethoxylate unit.

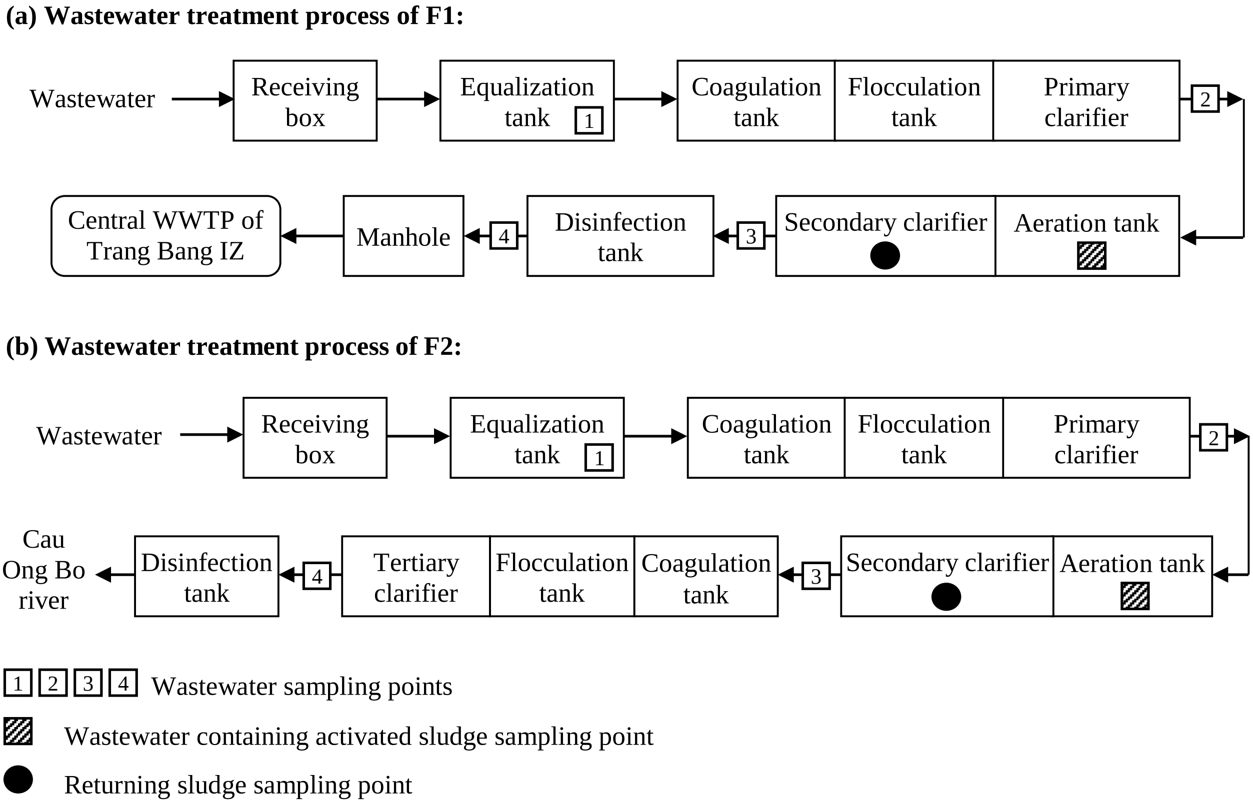

Figure 6.1 Wastewater treatment processes and sampling points at (a) Hoa Sen factory - F1 and (b) Chyang Sheng factory - F2

6.3.3 Method of analysis

Macro-compositions such as color, total suspended solids (TSS), mixed liquor suspended solids (MLSS), alkalinity, organic matter in terms of chemical oxygen demand (COD) and biological oxygen demand (BOD_5), ammonia ($N\text{-}NH_4^+$), nitrite ($N\text{-}NO_2^-$), and nitrate ($N\text{-}NO_3^-$) were analyzed at the laboratory of CENTEMA - Vietnam, following the standard methods for the examination of water and wastewater ([APHA] American Public Health Association 2005).

Solid phase extraction of NP and NPEOs was conducted at the Laboratory for Advanced Waste Treatment Technology of the National University of Ho Chi Minh city - Vietnam. The author followed the solid phase extraction procedures as described by Tuc Dinh et al. (2011). First, 200 mL of each wastewater sample was filtered through 0.7 μm glass fiber filters (GF/F, Whatman) to remove suspended solids. Second, the cartridges were conditioned with 3mL of high performance liquid chromatography grade methanol (MeOH) of Merck, and then equilibrated with 3 mL of Mili-Q water. SampliQ solid phase extraction cartridges C18 ODS (Agilent, USA) were used. Third, wastewater samples were loaded through the cartridges at the flow rate of 3 – 5 mL per minute. Depending on the characteristic of the wastewater samples, loading volumes were adjusted from 100 – 134 mL so that the cartridges would not get blocked. When the loading almost finished, the cartridges were rinsed with 3 mL of MeOH:Mili-Q (5:95) solution, and then vacuum dried for 10 minutes. Next, elution was performed with 5 mL of MeOH solution each. After that, evaporation of the extracts was conducted under a gentle 99.999% pure nitrogen stream at 40^oC to dryness. Residues were dissolved in 1 mL of gradient grade for liquid chromatography acetonitrile (CH_3CN) solution of Merck each and well shacked. Finally, the resulting solid phase extracts were injected through 0.45 μm syringe filters into 1.5 mL vials and preserved at 4^oC. The analysis of extracted samples was performed in Japan to take advantage of modern and more sensitive equipment needed for data accuracy. During the transportation to Japan, the samples were preserved in an ice container.

Analysis of NP and NPEOs was conducted at the laboratory of the Foundation for Promotion of Material Science and Technology of Japan. NP and NPEOs (from NP_1EO to $NP_{15}EO$) were quantified using a liquid chromatograph (LC – Prominence, Shimadzu, Japan) followed by a tandem mass spectrometer (MS - 4500 Qtrap, AB SCIEX, USA). In liquid chromatography, an Inertsil PH column (150 mm × 2.1 mm, 5 μm, GL Science, Japan) was employed and maintained at 40^oC, where the mobile phase A was a 5 mmol/L ammonium acetate (CH_3COONH_4) solution and the mobile phase B was a MeOH solution. The injection volume was 10 μL and the flow rate was at 0.2 mL per minute. In mass spectrometry, an electrospray ionization (ESI) method and a multiple reaction monitoring (MRM) mode were employed.

Recovery experiments were conducted with 100 mL filtered water samples spiked with 400 ng/L of each aforementioned alkylphenolic substances. The same solid phase extraction method and quantification method were applied.

6.4. Results and discussion

6.4.1 Design figures and operating conditions of the factories during the campaign

The design capacities of the WWTPs of F1 and F2 were 1,000 m^3/day and 3,500 m^3/day, respectively. Each factory had just resumed operations after a three-day holiday when the campaign began. During the campaign, the flowrate of F1 was at about 44% of its full capacity, and F2 was operating at approximately 54% of its full capacity, where the full flowrates were assumed as equal to 80% of the design capacity (a safety factor of 1.25). Operating figures of the biological process such as HRT in the aeration tank and SRT were achieved from the data of flowrates, aeration volumes, sludge returning scheme, and the analyzed results of MLSS, effluent solids concentration, and returning solids concentration. Technical figures of the two WWTPs that were necessary for the assessment of NPEO and NP removal were summarized in Table 6.2.

Table 6.2 Design and operating figures of the WWTPs of Hoa Sen and Chyang Sheng factories

Factory	Wastewater flowrate Q (m^3/day)		Aeration volume V_A (m^3)	MLSS/ MLVSS $X_A^{(a)}$ (mg/L)	Waste solids flow rate Q_W (m^3/d)	Returning solids concentration $X_R^{(a)}$ (mg/L)	Effluent solids concentration $X_E^{(a)}$ (mg/L)	HRT$^{(b)}$ (h)	SRT$^{(b)}$ (day)	R$^{(b)}$
F1	During the campaign	700	607.2	2,267/ 1,651	0.8	9,913	21	20.8	60.8	0.29
	At full capacity$^{(b)}$	800	607.2	-	-	-	-	18.2	-	-
F2	During the campaign	1,500	1,013.8	2,657/ 2,141	1.6	4,753	22	16.2	66.3	1.25
	At full capacity$^{(b)}$	2,800	1,013.8	-	-	-	-	8.7	-	-

Notes: MLSS/ MLVSS: mixed liquor suspended solids/ mixed liquor volatile suspended solids in aeration tank. HRT: Hydraulic retention time in the aeration tank. SRT: Solids retention time in the activated sludge process. R: returning activated sludge ratio. Correspondingly, returning sludge flowrates for F1 and F2 were 205 m^3/day and 1,881 m^3/day, respectively. $^{(a)}$ Analysis result; $^{(b)}$ Estimation by the authors; "-" means data not available.

6.4.2 Textile wastewater compositions

Wastewater released from manufacturing processes differs in both quantities and quality, depending on dyeing technologies as well as types and doses of chemicals (CENTEMA 2010). Combined streams from equalization tanks of various textile factories across Vietnam are characterized by high values of pH (over the favorable range for microbial growth as 8.5), temperature (36 – 52^{o}C), color (350 – 3,710 Pt-Co), organic matter expressed in COD (360 – 2,448 mgO_2/L) and BOD_5 (200 – 1,450 mgO_2/L) (CENTEMA 2010).

Influent wastewater of both factories was collected from equalization tanks with aeration where wastewater flowrates and compositions were homogenized. In comparison with the reported compositions and characteristics of textile wastewater in Vietnam, influent wastewater compositions of F1 and F2 (Table 6.3), were rather low with average TSS values of 139 and 159 mg/L, and COD of 317 and 541 mgO_2/L, respectively. Exceptionally, the wastewater had high color values of 848 Pt-Co (F1) and 945 Pt-Co (F2). The influent pH of F1 ranged from 8.6 up to 10.0, while that of F2 was moderately lower, between 7.9 and 8.6. As a characteristic of textile industry, the influent wastewater was high in temperature, at an average of 40^{o}C and 41^{o}C for F1 and F2, respectively. Comparing compositions of the disinfection effluent (F1) and the tertiary effluent[7] (F2) with the national technical regulation on the effluent of textile industry, QCVN 13:2008/BTNMT, it was suggested that only F1 might fully comply with the standard in terms of pH, color, TSS, and organic matter.

Table 6.3 Compositions of influent and effluent textile wastewater at the two factories

Composition	Unit	Hoa Sen factory – F1[(a)]		Chyang Sheng factory – F2[(a)]		QCVN 13:2008/BTNMT	
		Influent	Disinfection effluent	Influent	Tertiary effluent	Class A	Class B
pH	-	8.6 – 10.1	7.8 – 8.0	7.9 – 8.6	6.8 – 7.4	6 - 9	5.5 – 9.0
Temperature	°C	40	-	41	-	40	40
Color (pH = 7)	Pt-Co	848	31	945	87	20[(b)]/50[(c)]	150
TSS	mg/L	139	19	159	19	50	100
COD_{total}	mgO_2/L	317	-	541	-	50	150
COD_f	mgO_2/L	247	36	416	101	-	-
BOD_{5-f}	mgO_2/L	163	7	180	57	30[(d)]	50[(d)]

Notes: [(a)] Average compositions in the three-day campaign, this study. [(b)] For newly established factories; [(c)] For the existing factories. [(d)] Maximum allowable level for total BOD_5. COD_f and BOD_{5-f}: COD and BOD_5 of wastewater samples that had passed through 0.7 μm glass fiber filters, respectively. Class A: wastewater to be discharged into the waters for domestic supply. Class B: wastewater to be discharged into the waters for other purposes except for domestic supply. "-" means data not available.

In order to assess the removal of NPEOs and NP, wastewater samples at the effluents of each treatment stages were analyzed for both macro-pollutants and micro-pollutants as $NP_{1-15}EO$ and NP. The analysis results are presented in Table 6.4. The facilities demonstrated rather high efficacy in overall removing of COD_f (75% and 82%), BOD_{5-f} (95% and 69%), and TSS (effluent TSS of 18 and 19 mg/L) for F1 and F2, respectively. During the campaign, F2 might perform a partial nitrification. The ammonia removal efficiency was 23%, equivalent to 20 mg/L. The loss in ammonia might be attributed to the ammonia oxidation and the conversion of slowly biodegradable organic matter (a COD_f reduction of 73%) into new cells. Nitrite concentration showed to be accumulated with an additional amount of 8 mg/L. A small decrease in nitrate (0.6 mg/L) was probably due to the assimilation by heterotrophic bacteria or/and denitrification occurring in one section (one third) of the aeration tank where moving bed media was introduced (Yang et al. 2009, Houweling et al. 2007). However, ammonia was not detected from the primary and secondary effluents of F1. It was suggested that ammonia in the equalization tank of F1 probably evaporated at high pH (pH = 10.1), so a conclusion on nitrifying of F1 could not be made. Further discussion on nitrification conditions linked with NPEO and NP removal will be provided in the next section.

Regarding micro-pollutants, both NPEOs and NP were detected in the influent wastewater of the two factories. The concentration of total $NP_{1-15}EO$ of F1 was 117,594 ng/L, which was about 40 times higher than that of F2, as 3,099 ng/L. The presence of NP at 474 ng/L and 551 ng/L in the

[7] During the campaign, the disinfection of F2 was not in operation.

influents of F1 and F2, respectively, was due to the degradation of the parent ethoxylates in the sewage systems. Our study demonstrated rather good overall removals of NPEOs (at the rates of >99% and >79%), and NP (at the rates of 77% and 56%) for F1 and F2, respectively. NP concentrations of the secondary effluents were below the European Union environmental quality standard of 0.3 μg/L (European Parliament and Council 2008). The reported results of NP and NPEOs had been adjusted using recovery rate for individual compounds (Figure 6.2).

Table 6.4 Wastewater compositions after each treatment process

Composition	Unit	Hoa Sen factory[a]				Chyang Sheng factory[b]			
		Equalizat-ion	Primary effluent	Second -ary effluent	Disinfect -ion effluent	Equalizat-ion	Primary effluent	Second -ary effluent	Tertiary effluent
pH	-	10.1	6.7	7.7	7.9	8.1	7.1	7.8	7.4
Temperature	^oC	39	-	34[c]	-	41	-	37[c]	-
DO	mgO_2/L			4.8[c]				4.3[c]	
Alkalinity	mg $CaCO_3/L$	700	280	320	-	450	300	190	-
TSS	mg/L	46	25	25	18	212	3	15	19
COD_f	mgO_2/L	265	157	58	66	570	504	138	104
BOD_{5-f}	mgO_2/L	165	28	25	8	297	246	53	92
$N-NH_4^+$	mg/L	55	<MDL	<MDL	-	85	87	67	-
$N-NO_2^-$	mg/L	0.4	0.2	0.0	-	0.8	0.4	8.4	-
$N-NO_3^-$	mg/L	1.0	2.2	2.1	-	3.4	2.3	1.7	-
NP	ng/L	474	487	109	-	551	1,141	244	-
$\Sigma NP_{1-15}EO$	ng/L	117,594	6,469	<SQL	-	3,099	4,453	<SQL	-

Notes: [a] Wastewater compositions on September 6th, 2016; [b] Wastewater compositions on September 7th, 2016. [c] Parameters of wastewater in aeration tanks. MDL: Method detection limit; MDL_{NH4+} = 5 mg/L. SQL: Sample Quantification Limit; $SQL_{NPEOs-F1}$ = 540 ng/L; $SQL_{NPEOs-F2}$ = 660 ng/L. "-" means data not available.

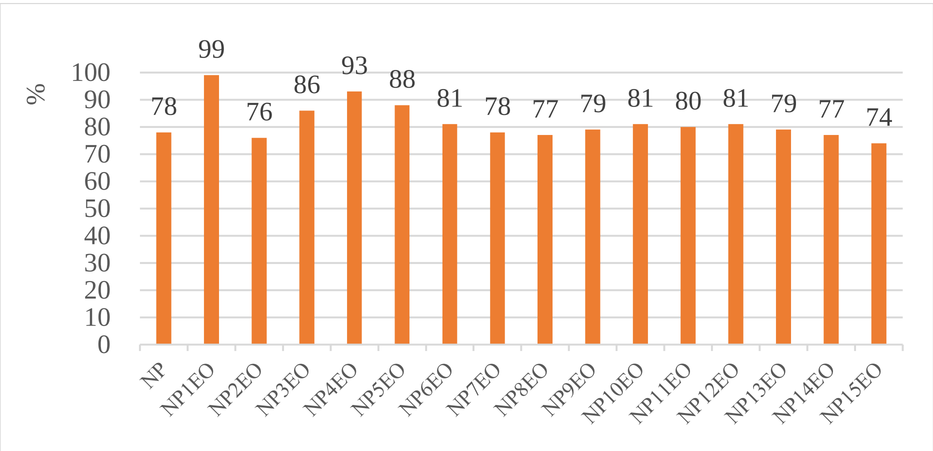

Figure 6.2 Recovery rate for nonylphenol and individual nonylphenol ethoxylate(s)

An inspection of oligomer distributions (Figure 6.3 and 6.4) revealed similar patterns for the influent wastewater and the primary effluents for both WWTPs. The influent wastewater from F1 (specializing in cotton materials) contained $NP_{4-17}EO$ in majority, accounting for 98.5% of the detected homologues, which seemed close to the distribution of NPEO oligomers found by Loos et al. (2007). Accordingly, the proportion of individual oligomers of this group varied between

3.0% and 13.7%. In contrast, influent wastewater from F2 which based on synthetic fabrics was dominated by short-chain oligomers (NP_1EO: 61.4% and NP_2EO: 12.4%), reflecting the similarity with the findings of Pothitou and Voutsa (2008). Because the total $NP_{1-15}EO$ levels of the secondary effluents were below the sample quantification limits for both factories, they were not shown in the following figures.

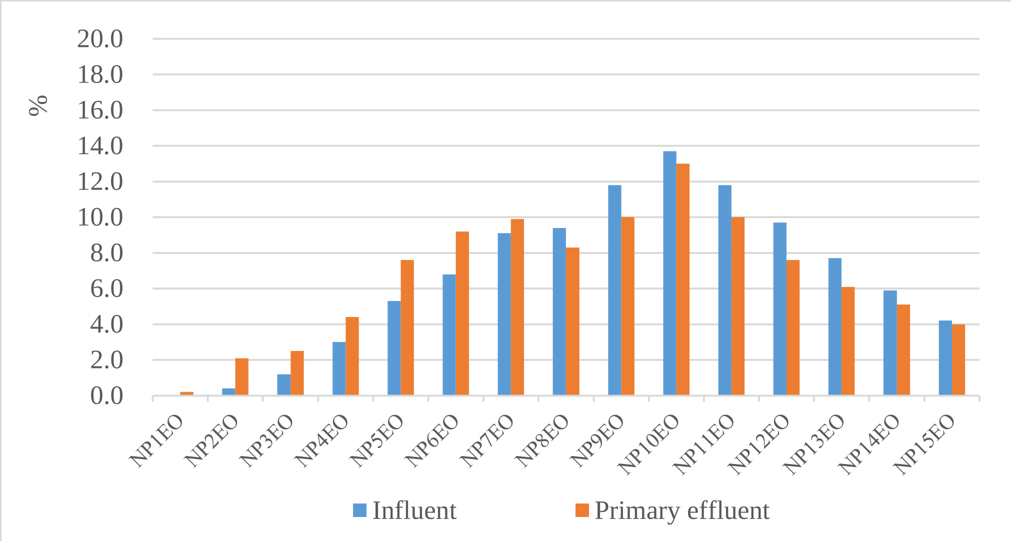

Figure 6.3 Distribution of nonylphenol ethoxylates for F1

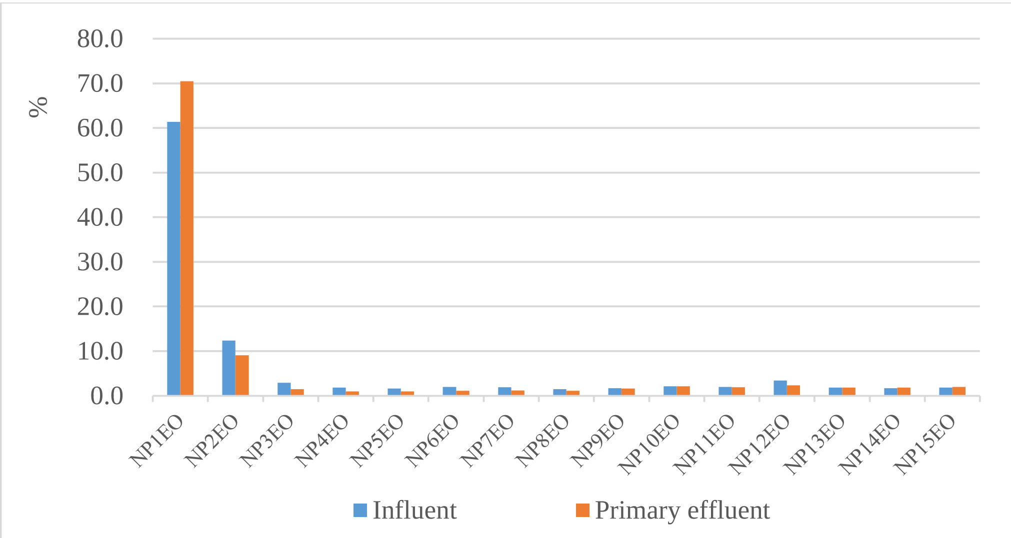

Figure 6.4 Distribution of nonylphenol ethoxylates for F2

6.4.3 NPEO and NP removal of the two textile factories of investigation

In this section, critical factors influencing the removal of NPEOs and NP by the wastewater treatment processes, which have been applied in the two textile factories, are addressed and discussed. With the achieved data of NPEOs and NP, the primary coagulation and the secondary ASP will be focused. Consequently, the effects of temperature and TSS in the primary coagulation, as well as the roles of HTR, nitrification, SRT, and MLSS in the ASP on NPEO and NP removal will be discussed in depth.

The role of primary coagulation

Coagulation has been widely applied as primary treatment (e.g. at Hoa Sen factory – F1) or as tertiary treatment or both (e.g. at Chyang Sheng factory – F2) in textile wastewater treatment in Vietnam (VEA 2011). It is capable of eliminating 73% to 85% of the influent TSS (CENTEMA 2010), and about 50% of the color (Loan 2011). Regarding nonylphenolic compounds, Stackelberg et al. (2007) and Nam et al. (2014) reported an NP removal of about 15 - 16% during the coagulation process. Notwithstanding, greater removals of 60% of nonylphenol equivalent (NP_{equ}) (Ahel et al. 1994a) or up to 75% of NP (Nam et al. 2014) could be obtained with an increased turbidity in terms of solids concentration. On the contrary, a shortening of medium-long NPEOs, which are more hydrophilic, to produce more persistent metabolites has been observed in primary treatment by Ahel et al. (1994a) and Loyo-Rosales et al. (2007a). This process is elevated with an increased wastewater temperature (Loyo-Rosales et al. 2007b).

Results from F1 revealed that $NP_{2\text{-}15}EO$ concentration was reduced by 94% from 117,594 ng/L to 6,469 ng/L during the primary coagulation process (Figure 6.5). At the same time, a small elevation of NP and NP_1EO by about 5%, was observed. The net increase in NP and NP_1EO suggests that their formation from the degradation of the parent oligomers ($NP_{2\text{-}15}EO$) slightly exceeds their removal via adsorption onto solids. This result , on the one hand, confirms the finding of Vogelsang et al. (2006) that the coagulation process primarily removes the most long-chain ethoxylates, where a temperature of 40^{o}C of the influent wastewater greatly contributes to the elimination of these compounds. On the other hand, it has reflected a high affinity of NP and NP_1EO on particulate matter due to their great hydrophobicity.

In F2 there was a 19% reduction in $NP_{3\text{-}7}EO$, from 337 ng/L to 281 ng/L, in relation to a 66% increase of the metabolites mainly as NP and NP_1EO, from 2,745 ng/L to 4,552 ng/L, during the primary coagulation process (Figure 6.6). At the same time, a small amount of $NP_{8\text{-}15}EO$ was produced, raising their concentration from 569 ng/L to 762 ng/L. As discussed earlier, the observed behaviors of NP and their ethoxylates have followed two main mechanisms as sorption and biotransformation suggested by Fauser et al. (2001). Particularly, our results have shown unsimilar patterns of transformation between intermediate oligomers ($NP_{3\text{-}7}EO$) and higher oligomers, which have been described by Ahel et al. (1994a). The increase in $NP_{8\text{-}15}EO$ level was probably due to the degradation of longer-chain oligomers. Indeed, higher oligomers with more than 15 ethoxylate groups such as $NP_{4\text{-}17}EO$ (Loos et al. 2007) and $NP_{35\text{-}40}EO$ (Vogelsang et al. 2006) have been reported for textile wastewater.

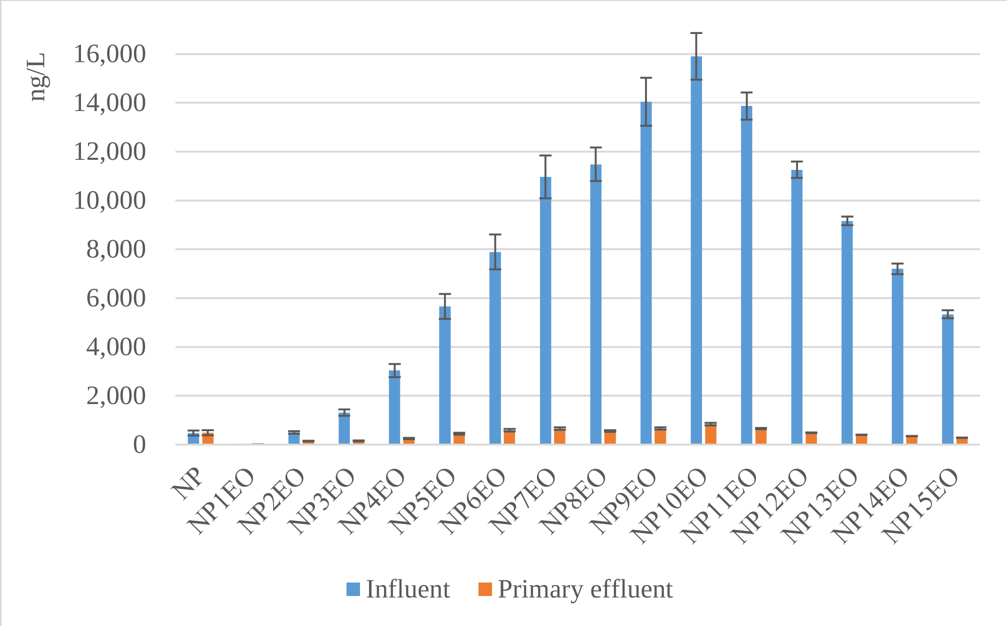

Figure 6.5 Concentration of NP and individual NP_nEO for F1

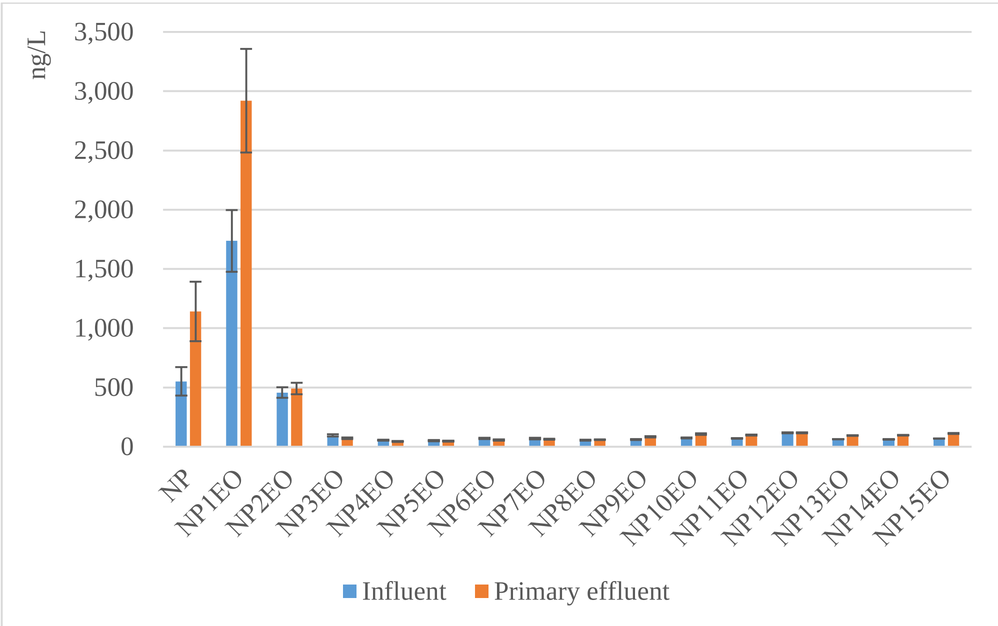

Figure 6.6 Concentration of NP and individual NP_nEO for F2

The role of secondary activated sludge process

- Removal efficacy of NPEOs and NP

Although anaerobic processes and advanced aerobic processes (e.g. membrane bioreactor – MBR, moving bed biofilm reactor – MBBR) have been introduced, conventional ASPs still dominate in textile wastewater treatment in Vietnam. Hence, discussing the removal efficacy of NPEOs and NP as well as influencing factors of this process is of great importance that contributes to the practice of wastewater treatment design and operation in Vietnamese textile industry.

Our study at Hoa Sen factory (F1) and Chyang Sheng factory (F2) revealed good removals of NP by the ASPs, at 78% and 79%, respectively. The estimation of removal efficacy did not take into account the formation of NP during the ASPs. The effluent concentrations of NP, as low as 109 ng/L (F1) and 244 ng/L (F2) are comparative with those from a similar treatment process reported by Vogelsang et al. (2006). Both facilities also showed satisfactory reductions of $NP_{1-15}EO$ to <540 ng/L (F1) and <660 ng/L (F2), equivalent to removal rates of >92% and >85%, respectively. Our results of NPEO removal are slightly lower than those achieved from ASPs (92 – 96%) reported by Petrie et al. (2014). It is possibly because the study of Petrie et al. (2014) is based on well-controlled pilot-scale models. The followings will discuss about possible operational conditions that may affect the NPEO and NP removal efficacies of F1 and F2.

- Persistence of NP and NPEO metabolites and the implication of HRT

In an activated sludge system designed for EDC removal, HRT is a parameter of great importance since a complete degradation can be achieved with an adequate HRT (Langford et al. 2007, Scruggs et al. 2004). The biotransformation of NPEOs occurs in two steps. Firstly, the oxidation of ethoxylate chain as primary degradation may take place as quickly as within 10 hours of aeration (Carvalho et al. 2000, Nimrod and Benson 1996). The next phase as mineralization of decomposed products may consume an additional 10 hours for the induction of specialized enzymes or shifts in bacterial inoculum that may degrade those substances via different mechanisms (Birkett and Lester 2003, Carvalho et al. 2000, Maki et al. 1994). Consequently, an ultimate degradation time of about 20 hours at the F/M (substrate/biomass) ratio of 0.24 – 0.86 is suggested by Carvalho et al. (2000). Nevertheless, Petrie et al. (2014) argue that shortening of medium-to-long nonylphenolics ($NP_{4-12}EO$) may not be dependent on the HRT, whereas the degradation of short-chain compounds ($NP_{1-4}EO$) and NP is probably mediated by increasing HTR from 8 to 24 hours.

During our campaign, the activated sludge processes of F1 and F2 were operated at HRT values of 21 hours and 16 hours, corresponding to operational capacities of 700 m^3/day and 1,500 m^3/day, respectively. High removals of NP and $NP_{1-15}EO$ obtained for both F1 and F2 were consistent with the findings of Petrie et al. (2014) as aforementioned. However, at full operations (F1: 800 m^3/day, F2: 2,800 m^3/day), HRT values would be shortened as 18 hours and 9 hours for F1 and F2, respectively. In that case, F1 may be still on the safe side regarding nonylphenolic removal, but F2 may be at risk in terms of completely mineralization of NPEO metabolites as suggested by Birkett and Lester (2003), Carvalho et al. (2000), Maki et al. (1994), and Petrie et al. (2014).

From experiences of design and operation of WWTPs in Vietnam, it is suggested that HRTs of between 6 and 8 hours would be sufficient for textile wastewater treatment (CENTEMA 2010). Taking into account of micro-pollutant removal, our study among others has provided evidences that HRTs of no less than 16 hours might be favorable for removing NP and its parent compounds from textile wastewater. Therefore, it is suggested that the existing ASPs with extended HRTs for textile wastewater treatment in Vietnam should be maintained. Nevertheless, long HRT systems may result in large footprints. Therefore, for those enterprises of space constraint who are likely to improve their WWTPs' performance, a modified approach is to provide carrier materials, specifically fixed-film, into activated sludge systems, where EDC removal could be enhanced as hydrophobic organic substances are trapped onto the surface of floc (Johnson and Darton 2003).

- The role of nitrification

It has been indicated that nitrifying conditions may play an important role in NP and short-chain NPEO removal with ASPs (Birkett and Lester 2003, Cirja et al. 2008, Loyo-Rosales et al. 2007b, Scruggs et al. 2004). At the same time, NPEOs with longer chain (more ethoxylate groups) are more eager to be decomposed. In nitrifying conditions, co-metabolic oxidation with ammonium monooxygenase enzyme possibly allows the assimilation of unreadily biodegradable EDCs (Cicek et al. 2001, Kim et al. 2007a, Kreuzinger et al. 2004). Körner et al. (2001) reported a marginal estrogenic response (RPE[8] = 10%) of the effluent from a textile WWTP adopting an ASP with nitrification-denitrification. In a strong agreement, Vader et al. (2000) suggested that high removal of estrogens (in the range of 79 - 95%) was mainly attributed to the nitrifying stage. Findings of McAdam et al. (2011) also showed the same patterns where nitrifying appeared in higher de-ethoxylation of long-chain nonylphenolic compounds ($NP_{5-12}EO$), at approximate 92%, and partly removal of NP, whereas non-nitrifying process yielded poorer de-ethoxylation and an accumulation of NP.

Conditions for nitrifying in wastewater treatment have been widely studied. It is suggested that nitrification could be affected by various factors, such as pH coupled with alkalinity, temperature, DO, organic loading, toxic compounds available in wastewater, HRT and SRT, which may vary between ammonium oxidizing bacteria (AOB) and nitrite oxidizing bacteria (NOB) (Gerardi 2005, Hanaki et al. 1990, Ito and Matsuo 1980, Sharma and Ahlert 1977, Zhu et al. 2008, Van Hulle et al. 2007, Li et al. 2013). Accordingly, alkaline pH of 8.4 – 8.5, temperature of 28 – 36^{o}C, and organic loading in the range of 0.15 – 1.1 gCOD/gMLVSS.day (Sharma and Ahlert 1977) were reported with uninhibited nitrification. A study of Park et al. (2007) provided insights of the optimal pH values of 8.2 ± 0.3 for AOB and 7.9 ± 0.4 for NOB. Increasing HRTs from 12 to 20 and to 22 hours exhibited a poor nitrification, a partial nitrification with peak nitrite accumulation, and a complete nitrification, respectively (Li et al. 2013).

Compared with the reported nitrifying conditions, the pH values in the ASPs of the two factories of 7.7 – 7.8 did not lie in the optimal range but might sustain the development of both AOB and NOB (Park et al. 2007). The pH in the ASP is partly influenced by the primary process where pH adjusting chemicals are introduced to facilitate the coagulation reactions. The DO levels of 4.8 and 4.3 mgO_2/L indicated an aerobic environment, and the temperatures of 34 and 37^{o}C might provide proper conditions for nitrifiers in F1 and F2, respectively. Our estimation of organic loadings

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[8] RPF: Relative proliferative effect. In the study of Körner et al. (2001), RPF indicated the proliferative effect of effluent wastewater on the MCF-7 breast cancer cells relative to the positive control 17β-estradiol (E2).

resulted in acceptable values for nitrification, at 0.1 gCOD/gMLVSS.day (F1) and 0.2 gCOD/gMLVSS.day (F2). An HRT of 20.8 hours might favor both nitritation and nitratation in F1, whereas an HRT of 16.2 hours in F2 was beneficial for the growth of AOB superior to NOB, leading to an accumulation of nitrite as aforementioned (Guo et al. 2009, Li et al. 2013).

Importantly, nitrification can take place in the presence of autotrophic bacterial species that are believed to be slowly growing (Luo et al. 2014, Sharma and Ahlert 1977). Thus, nitrifying usually involves an extended SRT that sustains the development of a diverse microbial population including the nitrifiers (Cirja et al. 2008, Loyo-Rosales et al. 2007b). The role of STR in NP and NPEO removal will be discussed in the next section.

- Extended SRT as an essence of microbial development and biomass quality

A complete biotransformation of short-chain NPEOs, as aforementioned, may require a specialized microbial inoculum that could be enriched in an elevated SRT system (Cirja et al. 2008, Kreuzinger et al. 2004, Scruggs et al. 2004, Terzic et al. 2005). According to Cirja et al. (2008), microbial communities including nitrifying bacteria are able to acclimate and develop at the SRTs of longer than 8 days. Langford et al. (2007) reported an SRT of 14 days which was sufficient to obtain a diverse population of microorganisms. The authors also indicated a greater production of shorter-chain NPEOs along with rapid losses of long-chain oligomers. Indeed, an investigation of three WWTPs in Beijing - China by Zhou et al. (2009) demonstrated the role of SRT. The results showed that the system with longer SRT (16 – 17 days) obtained a higher NP removal (65%) compared with the other two. Satisfying elimination of NP was achieved at the SRT of 25 days (Ivashechkin et al. 2004) up to 30 days (Johnson et al. 2005). It should be noticeable that the degradation of medium-long NPEOs may not be necessarily acquired by long SRTs, which may be only the case for short-chain oligomers ($NP_{1-4}EO$) and NP (Petrie et al. 2014).

In addition, sorption onto activated sludge flocs of hydrophobic nonylphenolic compounds could be enhanced at high SRTs (Langford et al. 2007). An extended sludge age may be concurrent with an increase in MLSS (Langford et al. 2005b), where the partitioning of NP between dissolved and particulate phases is attributed to the SS or MLSS concentration in wastewater (Brunner et al. 1988). Moreover, it is suggested that bacteria surface may become more hydrophobic and less negatively charged at high SRTs, resulting in more partition (Liao et al. 2001).

From the operating figures at F1 and F2 (Table 6.2) such as aeration volume, biomass concentration, waste solids generation and concentration, and effluent solids concentration, SRT values were estimated using the mass balance approach (Metcalf and Eddy 2004). As a result, SRT values for F1 and F2 of approximately 61 days and 66 days, respectively, exceed the expected SRT for short-chain NPEO decomposition. Therefore, it is suggested that the success in NPEO and NP removal at the two factories is attributed to satisfactory SRTs although the pH might be unsupportive for an optimal nitrifying.

In the majority of textile WWTPs in Vietnam, the parameter of SRT is not sufficiently taken into account. Information on SRTs was hardly derived from a survey in 120 textile WWTPs by VEA (2011). It was also indicated that small and medium enterprises paid insufficient attention on operating WWTPs as well as training responsible staffs. According to Aboobakar et al. (2013), most of the existing ASPs have been designed at mid-ranged conditions such as 10-day SRT and 8-hour HRT, which may not support a complete NPEO and NP removal.

- The implication of MLSS in removing hydrophobic compounds

ASPs have an advantage in the removal of nonylphenolic compounds due to the abundance of bio-solids as suggested by Brunner et al. (1988). Partition of nonylphenolic compounds in secondary sludge has been reported by various researchers. Petrie et al. (2014) documented 58 – 99% of NP, 68 – 91% of $NP_{1-3}EO$, and 77 – 85% of $NP_{4-12}EO$ in attachment to bio-solids at the SRT of 27 days and HRTs of 8 – 24 hours. Similarly, NP was found in association with activated sludge from 62% (Yamamoto et al. 2003) up to 93% (Keller et al. 2003), or in the range of 80 – 90% at 1,000 - 1,700 mgMLSS/L (Brunner et al. 1988). Consequently, sorption in addition to biodegradation made up high removals, up to 99% of NP and $NP_{1-2}EO$ at a 1,676 mgMLSS/L (Pothitou and Voutsa 2008), and over 99% of NP at 3,000 mgMLSS/L (Stasinakis et al. 2010). Indeed, adsorption of NP and NPEOs on solids may keep increasing till the critical micelle concentration (CMC[9]) is reached (Stechemesser and Dobiáš 2005). Hence, a certain MLSS level should be sustained for nonylphenolic removal, which probably depends on their concentration in wastewater.

Regarding the two factories of investigation, the average MLSS values of F1 and F2 during the campaign were 2,267 mg/L and 2,657 mg/L, respectively, which fell between the MLSS range suggested for complete-mixing ASPs as 1,500 – 4,000 mg/L (Metcalf and Eddy 2004). In combination with field observation at the secondary clarifiers and data of the effluent TSS concentration (as 19 mg/L - Table 3), it could be suggested that good removals of bio-solids were achieved. This may lead to a considerable reduction in NP and short-chain NPEOs by sorption at the detected MLSS levels of F1 and F2.

6.4.4 Limitations of this study

This study is not without shortcomings. First, grab sampling insufficiently reflects trends and “real” concentrations of wastewater from the two factories, which would lead to bias in the conclusion on consistently good nonylphenolic removals. Results could only reflect the concentrations and removal efficacies at the time of survey. Second, although the equalization tanks function in regulating flows as well as variations in wastewater compositions on a day, short-term variations in terms of hourly events could occur, leading to variations in NPEO and NP removal efficacies

[9] CMC is the concentration at which the system’s free energy can be reduced by the aggregation of the surfactant molecules into clusters with the hydrophobic groups located at the center of the cluster and hydrophilic head groups towards the solution.

which were still undiscovered by this study. Third, this study detected NPEOs and NP in Vietnamese textile wastewater only in the aqueous phase. This should be addressed by future investigations over the solid phase.

6.5. Conclusions

This study among others has demonstrated the presence and distribution of NPEOs and their final metabolite as NP in textile wastewater. To the best of our knowledge, this is the first study on textile wastewater regarding a wide range of NPEOs (from NP_0EO to $NP_{15}EO$) and their elimination across two typical textile wastewater treatment processes in Vietnam. The insights of NPEO and NP removal capacity as well as influencing factors in Vietnamese textile industry are crucial to improve the current wastewater treatment situation.

Our findings revealed diverse trends in the distribution of $NP_{1-15}EO$ in wastewater of the two selected factories. The highest concentrations centered at $NP_{9-11}EO$ for F1, which was specialized for cotton products. On the contrary, metabolites including $NP_{1-2}EO$ and NP dominated in wastewater from the synthetic fibers process, as F2. The trends showed insignificant changes from the influents to the primary effluents.

The primary coagulation of F1 functioned well in eliminating NP and $NP_{1-2}EO$ while a significant amount of medium-long oligomers, as much as 94%, was transformed into the metabolites. Whereas, a lower removal efficacy was obtained in F2, probably due to a continuing decomposition of more complex oligomers containing more than 15 ethoxylate groups.

ASPs shown to be crucial phases in reducing NPEOs and NP in textile wastewater, taking the advantages of long HRTs (as 21 and 16 hours), low organic loadings (as 0.1 and 0.2 gCOD/gMLVSS.day), and extended SRTs (as 61 and 66 days) which sustained the MLSS concentration of greater than 2,000 mg/L for both F1 and F2, respectively. Treatment conditions such as DO, temperature, organic loading, and SRT were likely suitable for nitrification. However, the pH of 7.8 and the HRT of 16 hours in F2 seemed not optimal for a complete nitrifying process.

Concerning the disadvantage of a prolonged HRT linked with a footprint shortage, future studies need to focus on optimizing nonylphenolic removal in conditions of Vietnam, taking into account the relationship between HRTs and alternatives such as BMR and attached-growth ASPs. Although satisfactory elimination of NPEOs and NP has been achieved, further research may extend the investigation for nonylphenol ethoxy carboxylates (NPECs), especially $NP_{1-2}EC$, since they can contribute to the NP concentration in the aquatic environment. Last but not least, additional investigations on both aqueous and solid phases are suggested to gain more insights of the fate of NPEOs and NP during textile wastewater treatment processes.

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CHAPTER 7

CONCLUSIONS

7.1 Propositions

This dissertation postulates the integration of risk scientist and non-scientist perspectives in EDSs risk management agenda by means of the PRAVE framework. Whilst findings with regards to the four objectives have been discussed and concluded in previous the chapters, this section claims the propositions of the dissertation that are subjected to EDSs risk decision-making supported by PRAVE.

Core findings:

- The public's awareness of EDSs is rather low and their judgments of EDSs risk are uncertainty-laden.
- There is non-significant difference in the patterns of risk perception and self-protection between pregnant women and young mothers and the remaining people.
- Perceived EDSs risk has no effect on the public's habit of riverine fish consumption; meanwhile it is demonstrated that consuming EDS contaminated riverine food products poses a health risk to a large population of people.
- Frequency of consuming EDS contaminated riverine food products is a key factor linked with health risk.
- Foreign regulations on EDSs restriction indirectly and positively impact the environment through changing the EDSs consumption and discharge behavior among the textile firms.
- There is a potency of continued use and discharge of EDSs into the environment among the domestic textile firms.
- Well-designed and controlled coagulation-activated sludge processes can yield promising NP/NPEOs removal for the textile wastewater.

Policy implications:

- Immediate actions in order to tackle EDSs risk need a kick-start in urban cities.
- Building capacity for communicating and managing EDSs risk among governmental and non-governmental institutions (e.g., education, research, and medical care organizations) is urgent.
- Integrating NP/NPEOs into the current standards for sewage and industrial effluent discharge and for surface water is recommended.

- Assessments and monitoring of wastewater treatment facilities with regards to EDSs removal are required.
- A social policy is needed for raising the awareness of EDSs and health risks among the public.
- Health risk education and medical care programs should pay attention on assisting pregnant women-young mothers in early preparation for pregnancy and in baby-care period.
- A research supporting policy is recommended for accelerating a comprehensive EDCs risk assessment in Vietnam.
- Long-term strategies should aim to restrict the use of EDSs in industrial manufactures and products.
- Supporting the private textile firms in terms of information on regulatory and market changes, exporting requirements, technological and chemical innovations is necessary.
- While the socio-economic development plan is provincial-based, the moral and legal responsibilities of the "polluter" towards the pollution "sufferers" in terms of administrational boundary should be recognized in the legal system of environmental protection of Vietnam.

Practical implications:

- EDCs risk management could be kick-started by applying the PRAVE framework which allows to design comprehensive strategies by mediating scientific and societal perspectives.
- EDSs risk decision-making is supported by sub-models of ecological-health risk evaluation, risk perception-behavior, and firm's perception-behavior. It is also supported with a guideline for dietary advice regarding riverine products consumption and a guideline for assessment of wastewater treatment facilities regarding NP/NPEOs removal.

Social-political implications:

Integrating social reflections of risk helps convey public concerns to relevant stakeholders, enhancing the quality and legitimacy of risk decision-making process in Vietnam.

The findings could be of interest to:

- Policy decision-makers who are seeking approaches to improve the quality of the environment and public heath in Vietnam.
- The city/ provincial authorities of environment, health, industry and trade, and risk managers who are willing to listen to the voice of the "sufferers" and wishing to implement a model of risk governance.
- Risk scientists and researchers who are interested in studying EDCs risk and in enhancing the effectiveness of risk communication.

- Health care practitioners who are seeking for EDSs-risk-related knowledge that may be useful for health advice.
- Practitioners who are seeking for a comprehensive guideline for assessing wastewater treatment facilities regarding EDSs removal, particularly in textile industry.

7.2 Recommendations for future research

Considering the limitations of this dissertation, suggestions for further studies are as follows:

- The impacts of EDSs exposure on the ecosystem are not examined under the scope of this work, which highlights further studies.
- This research presents a prototype model for evaluating ecological and health risk from consuming NP contaminated riverine food products from Can Giuoc river. Future research should extend to other urban rivers and substances, as well as to cover a wide range of food and drink. For big rivers, the use of sophisticated multimedia fate model (e.g., level IV) would be more realistic.
- Future research should extend the governing factors of EDSs risk perception to educational level and income. Regarding EDSs risk perception in relation to uncertainty perception, the roles of media coverage and public trust should be considered.
- Assessing the available wastewater treatment facilities of the potential industries with regards to EDSs removal efficiency is recommended with a focus on those discharging directly into the environment.

7.3 Contributions of the PRAVE framework and sub-models

This dissertation demonstrates the PRAVE as a framework of societal-physical-based risk decision-making. The framework realizes the idea of bringing together two complementary aspects of risk evaluation: the societal and physical reflections of risk. Useful findings and implications suggest that it could be a prototype for studying risk from diverse perspectives. Regarding practical application, the PRAVE provides a useful risk decision-making scheme, in particular for low risk and high scientific uncertainty situations.

Three decision-making supporting tools have been developed and being ready for use in further studies:

- The methodological framework for the studying of risk perception and health behavior could be suggested to solve the problem of EDSs contamination with confidence thanks to good consistency and reliability. It could be applicable for studies of environmental and health psychology as well as health promotion and education related to chemical contamination problems.

- The integrated model of ecological and health risk evaluation could be applied for other rivers (at local level) or substances. In this case, calibrations for initial hydrological, environmental, and biological conditions, or/and chemical characteristics are needed.
- The model of firm's perception and behavior is promising to support the study of how firms of diverse industries make decision when they face dilemmas. It could also be useful for studying the feasibility and effectiveness of environmental policy at micro (firm/ industry/ community/ household) level.

APPENDIX 1

Contents:

Table A1.1 Construct reliability and validity (*n* = 331)

Latent construct	CR	AVE	MSV	ASV
General awareness of water pollution and reproductive health problems (GA)	0.9	0.6	0.2	0.1
Awareness of the pathways of exposure that may affect reproductive health (AP)	0.9	0.8	0.2	0.1
Specific awareness of endocrine disrupting surfactants (SA)	0.8	0.5	0.4	0.1
Risk belief (RB)	0.8	0.7	0.4	0.2
Risk concern (RC)	0.8	0.7	0.2	0.1
Perceived uncertainty (UN)	0.9	0.8	0.2	0.1
Risk acceptability (RAC)	0.9	0.7	0.3	0.1
Non-diet-related self-protective response (NDSP)	0.8	0.6	0.2	0.1

Table A1.2 Construct discriminant validity (*n* = 331)

Latent construct	Pearson correlations							
	GA	AP	SA	RB	RC	UN	RAC	NDSP
General awareness of water pollution and reproductive health problems (GA)	0.742							
Awareness of the pathways of exposure that may affect reproductive health (AP)	0.331**	0.887						
Specific awareness of endocrine disrupting surfactants (SA)	-0.078	0.014	0.707					
Risk belief (RB)	0.249**	0.225**	0.529**	0.814				
Risk concern (RC)	0.337**	0.057	0.184**	0.134*	0.850			
Perceived uncertainty (UN)	-0.009	-0.084	-0.400**	-0.317**	-0.403**	0.878		
Risk acceptability (RAC)	0.427**	0.029	0.084	0.478**	0.338**	0.066	0.857	
Non-diet-related self-protective response (NDSP)	0.204**	0.191**	0.357**	0.472**	0.137*	-0.210**	0.520**	0.755

Notes: * Correlation is significant at the 0.05 level (2-tailed); ** Correlation is significant at the 0.01 level (2-tailed). The square roots of the average variance extracted values were showed in shaded boxes.

Table A1.3 Multicollinearity statistics of independent constructs (n = 328)

Independent construct	Tolerance	VIF
General awareness of water pollution and reproductive health problems (GA)	0.68	1.5
Awareness of the pathways of exposure that may affect reproductive health (AP)	0.86	1.2
Specific awareness of endocrine disrupting surfactants (SA)	0.61	1.6
Risk belief (RB)	0.60	1.7
Risk concern (RC)	0.70	1.4
Perceived uncertainty (UN)	0.69	1.4

Table A1.4 Results of normality test for the constructs (n = 328)

Construct	Group	Minimum	Maximum	Mean	Standard deviation	Skewness			Kurtosis		
						Statistic	S.E.	z-value	Statistic	S.E.	z-value
General awareness of water pollution and reproductive health problems (GA)	Inexperienced	1.4	6.5	5.4	0.05	-2.4	0.16	-15.0	8.9	0.32	27.8
	Experienced	1.2	6.4	5.2	0.10	-2.2	0.25	-8.8	5.4	0.49	11.0
	N_Pr.Ym	1.2	6.5	5.4	0.06	-2.4	0.17	-14.1	7.4	0.33	22.4
	Pr.Ym	1.4	6.4	5.4	0.07	-2.0	0.22	-9.1	7.1	0.44	16.1
Awareness of the pathways of exposure that may affect reproductive health (AP)	Inexperienced	0.5	5.6	3.8	0.06	-0.6	0.16	-3.8	0.8	0.32	2.5
	Experienced	0.9	5.5	3.7	0.09	-0.7	0.25	-2.8	1.0	0.49	2.0
	N_Pr.Ym	0.5	5.6	3.7	0.06	-0.6	0.17	-3.5	0.9	0.33	2.7
	Pr.Ym	0.9	5.4	3.8	0.08	-0.4	0.22	-1.8	0.4	0.44	0.9
Specific awareness of endocrine disrupting surfactants (SA)	Inexperienced	0.1	3.5	1.8	0.06	-0.2	0.16	-1.3	-1.1	0.32	-3.4
	Experienced	0.1	3.5	1.7	0.09	0.3	0.25	1.2	-0.9	0.49	-1.8
	N_Pr.Ym	0.1	3.5	1.8	0.06	-0.1	0.17	-0.6	-1.1	0.33	-3.3
	Pr.Ym	0.3	3.5	1.8	0.08	0.1	0.22	0.5	-1.2	0.44	-2.7
Risk belief (RB)	Inexperienced	1.7	6.0	4.1	0.05	0.2	0.16	1.3	-0.3	0.32	-0.9
	Experienced	1.3	5.6	3.8	0.10	-0.3	0.25	-1.2	-0.3	0.49	-0.6
	N_Pr.Ym	1.3	6.0	4.1	0.06	-0.1	0.17	-0.6	0.3	0.33	0.9
	Pr.Ym	1.7	5.8	4.0	0.08	-0.1	0.22	-0.5	-0.3	0.44	-0.7
Risk concern (RC)	Inexperienced	0.7	4.9	4.0	0.06	-1.4	0.16	-8.8	1.8	0.32	5.6
	Experienced	0.8	4.9	4.0	0.09	-1.4	0.25	-5.6	1.8	0.49	3.7
	N_Pr.Ym	0.7	4.9	4.0	0.07	-1.5	0.17	-8.8	1.7	0.33	5.2
	Pr.Ym	2.0	4.9	4.1	0.07	-0.9	0.22	-4.1	-0.4	0.44	-0.9
Perceived uncertainty (UN)	Inexperienced	0.7	4.1	2.0	0.06	0.6	0.16	3.8	-0.7	0.32	-2.2
	Experienced	0.6	4.1	2.1	0.09	0.4	0.25	1.6	-0.7	0.49	-1.4
	N_Pr.Ym	0.7	4.1	2.1	0.07	0.4	0.17	2.4	-1.0	0.33	-3.0
	Pr.Ym	0.6	4.0	1.9	0.08	0.7	0.22	3.2	-0.1	0.44	-0.2
Risk acceptability (RAC)	Inexperienced	2.6	6.9	5.6	0.05	-1.0	0.16	-6.3	1.3	0.32	4.1
	Experienced	1.5	6.8	5.3	0.12	-1.3	0.25	-5.2	1.1	0.49	2.2
	N_Pr.Ym	2.6	6.9	5.5	0.06	-1.1	0.17	-6.5	1.1	0.33	3.3
	Pr.Ym	1.5	6.9	5.5	0.09	-1.8	0.22	-8.2	3.6	0.44	8.2
Non-diet-related self-protective response (NDSP)	Inexperienced	2.5	6.5	5.5	0.05	-0.8	0.16	-5.0	0.6	0.32	1.9
	Experienced	1.8	6.5	5.2	0.11	-0.8	0.25	-3.2	0.3	0.49	0.6
	N_Pr.Ym	2.3	6.5	5.4	0.06	-0.8	0.17	-4.7	0.4	0.33	1.2
	Pr.Ym	1.8	6.5	5.5	0.08	-1.1	0.22	-5.0	2.1	0.44	4.8
Diet-related self-protective response (DSP)	Inexperienced	1.0	5.0	2.7	0.07	0.1	0.16	0.6	-0.6	0.32	-1.9
	Experienced	1.0	5.0	3.0	0.11	0.0	0.25	0.0	-0.6	0.49	-1.2
	N_Pr.Ym	1.0	5.0	2.8	0.08	0.0	0.17	0.0	-0.6	0.33	-1.8
	Pr.Ym	1.0	5.0	2.7	0.11	0.2	0.22	0.9	-0.6	0.44	-1.4

Note: Inexperienced: lay public; Experienced: experienced public; Pr.Ym: pregnant women and young mothers; N_Pr.Ym: the remaining population.

APPENDIX 2

Contents:

Table A2.1 Chemical concentration in the upstream canals

Parameter	Symbol	Unit	Value	Reference/ Equation
Average NP concentration in dissolved phase	**$C_{0\text{-}w}$**	µg/L	**9.70**	Hanh (2015)
Volatile organic matter fraction	f_{vom}	%	9.1	Average estimate from Hanh (2015)
Sediment-water partition coefficient	$K_{0\text{-}sed/spm}$	L/kg	344.2	
Organic carbon fraction	POC	%	5.1	POC = $0.56f_{vom}$ (Mackay 2001)
Organic carbon-normalized partition coefficient	K_{oc}	L/kg	6,754	$K_{oc} = K_{0\text{-}sed}/POC$
	Log K_{oc}	L/kg	3.83	
NP concentration in SPM (estimated)	**$C_{0\text{-}spm}$**	ng/g dw	**3,338**	$C_{0\text{-}spm} = C_{0\text{-}w}K_{0\text{-}spm} = C_{0\text{-}w}K_{oc}POC$
NP concentration in sediment (estimated)	**$C_{0\text{-}sed}$**	ng/g dw	**3,338**	$C_{0\text{-}sed} = C_{0\text{-}w}K_{0\text{-}sed} = C_{0\text{-}w}K_{oc}POC$

Table A2.2 Water quality of the Can Giuoc river

Parameter	Unit	Value	Reference
pH	-	6.7	SIFEP (2013)
DO	mgO_2/L	2.2	
SPM	mg/L	171	
POC	%	4.0	Minh et al. (2007)

Table A2.3 Chemical properties: 4-Nonylphenol (branched)

Properties	Symbol	Unit	Value	Reference/ Equation
CAS No.	-	-	84852-15-3	EC (2002)
Molecular formula	-	-	$C_{15}H_{24}O$	
Molecular weight	M	g/mole	220.34	
Vapor pressure	P^s	Pa at 25°C	0.30	
Water solubility	C^s	mol/m^3 at 20°C & pH7	0.027	
Henry's law constant	H	Pa.m^3/mol	11.0	$H = P^s/C^s$
n-Octanol-water partition coefficient	Log K_{ow}	L/kg	4.48	Ahel et al. (1993)
Biodegradation half-life in water	$t^w_{1/2}$	day	14	QSAR, Expert Survey (USEPA 2014)
Biodegradation half-life in water	$t^w_{1/2}$	h	336	QSAR, Expert Survey (USEPA 2014)
Biodegradation rate in water	k_w	1/h	0.002	$k_w = 0.693/t^w_{1/2}$
Biodegradation half-life in sediment	$t^{sed}_{1/2}$	day	49.5	Yuan et al. (2004)
	$t^{sed}_{1/2}$	h	1,188	
Biodegradation rate in sediment	k_{sed}	1/h	0.0006	$k_{sed} = 0.693/t^{sed}_{1/2}$

Table A2.4 Solids-water partition coefficients and Z values

Parameter	Symbol	Unit	Value	Reference/ Equation
Organic carbon fraction of SPM	POC_{spm}	%	4.0	Minh et al. (2007)
Organic carbon fraction of sediment	POC_{sed}	%	4.0	
SPM-water partition coefficient	K_{spm}	L/kg	272.4	$K_{spm} = K_{oc}POC_{spm}$
Sediment-water partition coefficient	K_{sed}	L/kg	272.4	$K_{sed} = K_{oc}POC_{sed}$
Dry bulk density of SPM	ρ_{spm}	kg/L	1.20	Xue et al. (2010)
Dry bulk density of surface sediment	ρ_{sed}	kg/L	1.20	
Fugacity capacity of NP in water	Z_w	mol/m^3.Pa	0.09	$Z_w = 1/H$
Fugacity capacity of NP in SPM	Z_{spm}	mol/m^3.Pa	29.7	$Z_{spm} = K_{spm}\rho_{spm}/H$
Fugacity capacity of NP in sediment	Z_{sed}	mol/m^3.Pa	29.7	$Z_{sed} = K_{sed}\rho_{sed}/H$

Medium scenario

The evaluative environment

Parameter	Symbol	Unit	Value	Reference/ Equation
Average width	W	m	171.3	Outcome of hydrological model Mike 11
Average water depth	H_w	m	10.7	
Water volume	V_w	m^3	3.72E+07	Estimated from the river geographical characteristics
Water-sediment interface area	A	m^2	3.52E+06	
River flow rate (inflow = outflow)	G	m^3/h	7.470E+04	Low flow rate scenario
SPM flux	G_{spm}	m^3/h	10.63	$G_{spm} = SPM*G_w/\rho_{spm}10^6$
Depth of surface sediment layer	H_{sed}	m	0.05	Mackay (2001)
Dry bulk sediment volume	V_{sed}	m^3	9.68E+04	$V_{sed} = AH_{sed}(1-0.45)$
Water side mass transfer coefficient	k_{m-w}	m/h	1.92E-02	Zhang et al. (2011)
Sediment side mass transfer coefficient	k_{m-sed}	m/h	5.05E-05	Zhang et al. (2011)

Estimations of D values

Parameter	Symbol	Unit	Value	Reference/ Equation
Advection rate via bulk water	D_{Aw}	mol/h.Pa	6,780.4	$D_{Aw} = GZ_w$
Advection rate via SPM	D_{Aspm}	mol/h.Pa	315.5	$D_{Aspm} = G_{spm}Z_{spm}$
Diffusion rate between water-sediment	D_{Dw-sed}	mol/h.Pa	2,835.5	$D_{Dw-sed} = 1/[(1/k_{m-w}AZ_w) + (1/k_{m-sed}AZ_{sed})]$
Biodegradation rate in water	D_{Rw}	mol/h.Pa	6,972.6	$D_{Rw} = V_wZ_wk_w$
Biodegradation rate in sediment	D_{Rsed}	mol/h.Pa	1,675.1	$D_{Rsed} = V_{sed}Z_{sed}k_{sed}$

Emission of NP from municipal wastewater

Parameter	Symbol	Unit	Value	Reference/ Equation
Emission rate	N	mol/h	**3.48**	Based on constant rate injection method (De Doncker 2009): $N = C_{0-w}G + C_{0-spm}G_{spm}$

Initial chemical concentration and fugacity

Parameter	Symbol	Unit	Value	Reference/ Equation
NP concentration in dissolved phase	$C_{0\text{-}w}$	mol/m^3	4.40E-05	$C_{0\text{-}w}(mol/m^3) = C_{0\text{-}w}(\mu g/L)\ 10^{-3}/M$
NP concentration in SPM	$C_{0\text{-}spm}$	mol/m^3	1.82E-02	$C_{0\text{-}spm}(mol/m^3) = C_{0\text{-}spm}(ng/g)\ \rho_{spm}10^{-3}/M$
NP concentration in sed	$C_{0\text{-}sed}$	mol/m^3	1.82E-02	$C_{0\text{-}sed}(mol/m^3) = C_{0\text{-}sed}(ng/g)\ \rho_{sed}10^{-3}/M$
Initial fugacity in water	$f_{0\text{-}w}$	Pa	4.85E-04	$f_{0\text{-}w} = C_{0\text{-}w}/Z_w$
Initial fugacity SPM	$f_{0\text{-}spm}$	Pa	6.13E-04	$f_{0\text{-}spm} = C_{0\text{-}spm}/Z_{spm}$
Initial fugacity sediment	$f_{0\text{-}sed}$	Pa	6.13E-04	$f_{0\text{-}sed} = C_{0\text{-}sed}/Z_{sed}$
Ratio of initial fugacity	$f_{0\text{-}w}/f_{0\text{-}sed}$	-	0.79	or $f_w = 0.79f_{sed}$

Fugacity, concentration, and amount of NP in the evaluative environmental compartments

Parameter	Symbol	Unit	Value	Reference/ Equation
Fugacity of NP in sediment	f_{sed}	Pa	2.60E-04	$f_{sed} = N/\Sigma D$
Fugacity of NP in water	f_w	Pa	2.06E-04	$f_w = 0.79f_{sed}$
Fugacity of NP in SPM	f_{spm}	Pa	2.60E-04	$f_{spm} = f_{sed}$
Equilibrium concentration in water	C_w	mol/m^3	1.87E-05	$C_w = Z_wf_w$
	C_w	µg/L	**4.11**	$C_w(\mu g/L) = C_w(mol/m^3)M(g/mol)*10^3$
Equilibrium concentration in SPM	C_{spm}	mol/m^3	7.71E-03	$C_{spm} = Z_{spm}f_{spm}$
	C_{spm}	ng/g dw	**1,120**	$C_{spm} = K_{spm}C_w$
Equilibrium concentration in sediment	C_{sed}	mol/m^3	7.71E-03	$C_{sed} = Z_{sed}f_{sed}$
	C_{sed}	ng/g dw	**1,120**	$C_{sed} = K_{sed}C_w$
Amount of NP in water	m_w	mol	6.95E+02	$m_w = C_w(mol/m^3)\ V_w$
Amount of NP in SPM	m_{spm}	mol	4.09E+01	$m_{spm} = C_{spm}(mol/m^3)\ V_wSPM/10^6\rho$
Amount of NP in sediment	m_{sed}	mol	7.46E+02	$m_{spm} = C_{sed}(mol/m^3)\ V_{sed}$
Total amount	$\sum m$	**mol**	**1.48E+03**	

Mass-balance at equilibrium

Parameter	Symbol	Unit	Value	Reference/ Equation
Advection via bulk water	N_{Aw}	mol/h	1.39	$N_{Aw} = D_{Aw}f_w$
Advection via bulk SPM	N_{Aspm}	mol/h	0.06	$N_{Aspm} = D_{Aspm}f_w$
Diffusion between water & sediment	$N_{Dw\text{-}sed}$	mol/h	0.15	$N_{Dw\text{-}sed} = D_{Dw\text{-}sed}(f_{sed} - f_w)$
Biodegradation in bulk water	N_{Rw}	mol/h	1.43	$N_{Rw} = D_{Rw}f_w$
Biodegradation in sediment	N_{Rsed}	mol/h	0.44	$N_{Rsed} = D_{Rsed}f_{sed}$
Total amount	**N**	**mol/h**	**3.48**	

Residence time

Parameter	Symbol	Unit	Value	Reference/ Equation
Advection residence time	τ_A	day	38.3	$\tau_A = \sum m/(N_{Aw} + N_{Aspm} + N_{Dw\text{-}sed})/24$
Reaction residence time	τ_R	day	33.0	$\tau_R = \sum m/(N_{Rw} + N_{Rsed})/24$
Overall residence time	τ	day	17.7	$\tau = \sum m/N/24$

Low scenario

The evaluative environment

Parameter	Symbol	Unit	Value	Reference/ Equation
Average width	W	m	171.3	Outcome of hydrological model Mike 11
Average water depth	H_w	m	10.7	
Water volume	V_w	m^3	3.72E+07	Estimated from the river geographical characteristics
Water-sediment interface area	A	m^2	3.52E+06	
River flow rate (inflow = outflow)	G	m^3/h	4.727E+04	Low flow rate scenario
SPM flux	G_{spm}	m^3/h	6.728	$G_{spm} = SPM*G_w/\rho_{spm}10^6$
Depth of surface sediment layer	H_{sed}	m	0.05	Mackay (2001)
Dry bulk sediment volume	V_{sed}	m^3	9.68E+04	$V_{sed} = AH_{sed}(1-0.45)$
Water side mass transfer coefficient	k_{m-w}	m/h	1.92E-02	Zhang et al. (2011)
Sediment side mass transfer coefficient	k_{m-sed}	m/h	5.05E-05	Zhang et al. (2011)

Estimations of D values

Parameter	Symbol	Unit	Value	Reference/ Equation
Advection rate via bulk water	D_{Aw}	mol/h.Pa	4,290.5	$D_{Aw} = GZ_w$
Advection rate via SPM	D_{Aspm}	mol/h.Pa	199.6	$D_{Aspm} = G_{spm}Z_{spm}$
Diffusion rate between water-sediment	D_{Dw-sed}	mol/h.Pa	2,835.5	$D_{Dw-sed} = 1/[(1/k_{m-w}AZ_w) + (1/k_{m-sed}AZ_{sed})]$
Biodegradation rate in water	D_{Rw}	mol/h.Pa	6,972.6	$D_{Rw} = V_wZ_wk_w$
Biodegradation rate in sediment	D_{Rsed}	mol/h.Pa	1,675.1	$D_{Rsed} = V_{sed}Z_{sed}k_{sed}$

Emission of NP from municipal wastewater

Parameter	Symbol	Unit	Value	Reference/ Equation
Emission rate	N	mol/h	**3.48**	This emission rate is based on medium scenario and fixed for low and high scenarios. $f_w/f_{spm} = 0.79$ (assumed as a constant since the evaluative environment is at equilibrium) $N = C_{0\text{-}w}G + C_{0\text{-}spm}G_{spm}$ $= Z_w f_{0\text{-}w}G + Z_{spm}f_{0\text{-}spm}G_{spm}$ $= f_{0\text{-}w}(Z_wG + 1.25Z_{spm}G_{spm})$

Initial fugacity and chemical concentration

Parameter	Symbol	Unit	Value	Reference/ Equation
Initial fugacity in water	$f_{0\text{-}w}$	Pa	7.67E-04	$f_{0\text{-}w} = N/(Z_wG + 1.25Z_{spm}G_{spm})$
Initial fugacity SPM	$f_{0\text{-}spm}$	Pa	9.69E-04	$f_{0\text{-}spm} = f_{0\text{-}w}/0.79$
Initial fugacity sediment	$f_{0\text{-}sed}$	Pa	9.69E-04	$f_{0\text{-}sed} = f_{0\text{-}w}/0.79$
Ratio of initial fugacity	$f_{0\text{-}w}/f_{0\text{-}sed}$	-	0.79	or $f_w = 0.79f_{sed}$
NP concentration in dissolved phase	$C_{0\text{-}w}$	mol/m^3	6.96E-05	$C_{0\text{-}w} = Z_wf_{0\text{-}w}$
NP concentration in SPM	$C_{0\text{-}spm}$	mol/m^3	2.87E-02	$C_{0\text{-}spm} = Z_{spm}f_{0\text{-}spm}$
NP concentration in sediment	$C_{0\text{-}sed}$	mol/m^3	2.87E-02	$C_{0\text{-}sed} = Z_{sed}f_{0\text{-}sed}$

Fugacity, concentration, and amount of NP in the evaluative environmental compartments

Parameter	Symbol	Unit	Value	Reference/ Equation
Fugacity of NP in sediment	f_{sed}	Pa	3.07E-04	$f_{sed} = N/\Sigma D$
Fugacity of NP in water	f_w	Pa	2.43E-04	$f_w = 0.79f_{sed}$
Fugacity of NP in SPM	f_{spm}	Pa	3.07E-04	$f_{spm} = f_{sed}$
Equilibrium concentration in water	C_w	mol/m^3	2.21E-05	$C_w = Z_wf_w$
	C_w	µg/L	**4.86**	$C_w(\mu g/L) = C_w(mol/m^3)M(g/mol)*10^3$
Equilibrium concentration in SPM	C_{spm}	mol/m^3	9.11E-03	$C_{spm} = Z_{spm}f_{spm}$
	C_{spm}	ng/g dw	**1,324**	$C_{spm} = K_{spm}C_w$

Parameter	Symbol	Unit	Value	Reference/ Equation
Equilibrium concentration in sediment	C_{sed}	mol/m^3	9.11E-03	$C_{sed} = Z_{sed}f_{sed}$
	C_{sed}	ng/g dw	**1,324**	$C_{sed} = K_{sed}C_w$
Amount of NP in water	m_w	mol	8.22E+02	$m_w = C_w(mol/m^3)\ V_w$
Amount of NP in SPM	m_{spm}	mol	4.83E+01	$m_{spm} = C_{spm}(mol/m^3)\ V_wSPM/10^6\rho$
Amount of NP in sediment	m_{sed}	mol	8.82E+02	$m_{spm} = C_{sed}(mol/m^3)\ V_{sed}$
Total amount	**$\sum m$**	**mol**	**1.75E+03**	

Mass-balance at equilibrium

Parameter	Symbol	Unit	Value	Reference/ Equation
Advection via bulk water	N_{Aw}	mol/h	1.04	$N_{Aw} = D_{Aw}f_w$
Advection via bulk SPM	N_{Aspm}	mol/h	0.05	$N_{Aspm} = D_{Aspm}f_w$
Diffusion between water & sediment	N_{Dw-sed}	mol/h	0.18	$N_{Dw-sed} = D_{Dw-sed}(f_{sed} - f_w)$
Biodegradation in bulk water	N_{Rw}	mol/h	1.69	$N_{Rw} = D_{Rw}f_w$
Biodegradation in sediment	N_{Rsed}	mol/h	0.51	$N_{Rsed} = D_{Rsed}f_{sed}$
Total amount	**N**	**mol/h**	**3.48**	

Residence time

Parameter	Symbol	Unit	Value	Reference/ Equation
Advection residence time	τ_A	day	57.3	$\tau_A = \sum m/(N_{Aw} + N_{Aspm} + N_{Dw-sed})/24$
Reaction residence time	τ_R	day	33.0	$\tau_R = \sum m/(N_{Rw} + N_{Rsed})/24$
Overall residence time	τ	day	21.0	$\tau = \sum m/N/24$

High scenario

The evaluative environment

Parameter	Symbol	Unit	Value	Reference/ Equation
Average width	W	m	171.3	Outcome of hydrological model Mike 11
Average water depth	H_w	m	10.7	
Water volume	V_w	m^3	3.72E+07	Estimated from the river geographical characteristics
Water-sediment interface area	A	m^2	3.52E+06	
River flow rate (inflow = outflow)	G	m^3/h	1.336E+05	Low flow rate scenario
SPM flux	G_{spm}	m^3/h	19.02	$G_{spm} = SPM*G_w/\rho_{spm}10^6$
Depth of surface sediment layer	H_{sed}	m	0.05	Mackay (2001)
Dry bulk sediment volume	V_{sed}	m^3	9.68E+04	$V_{sed} = AH_{sed}(1-0.45)$
Water side mass transfer coefficient	k_{m-w}	m/h	1.92E-02	Zhang et al. (2011)
Sediment side mass transfer coefficient	k_{m-sed}	m/h	5.05E-05	Zhang et al. (2011)

Estimations of D values

Parameter	Symbol	Unit	Value	Reference/ Equation
Advection rate via bulk water	D_{Aw}	mol/h.Pa	12,126.4	$D_{Aw} = GZ_w$
Advection rate via SPM	D_{Aspm}	mol/h.Pa	564.2	$D_{Aspm} = G_{spm}Z_{spm}$
Diffusion rate between water-sediment	D_{Dw-sed}	mol/h.Pa	2,835.5	$D_{Dw-sed} = 1/[(1/k_{m-w}AZ_w) + (1/k_{m-sed}AZ_{sed})]$
Biodegradation rate in water	D_{Rw}	mol/h.Pa	6,972.6	$D_{Rw} = V_wZ_wk_w$
Biodegradation rate in sediment	D_{Rsed}	mol/h.Pa	1,675.1	$D_{Rsed} = V_{sed}Z_{sed}k_{sed}$

Emission of NP from municipal wastewater

Parameter	Symbol	Unit	Value	Reference/ Equation
Emission rate	N	mol/h	**3.48**	This emission rate is based on medium scenario and fixed for low and high scenarios. f_w/f_{spm} = 0.79 (assumed as a constant since the evaluative environment is at equilibrium) $N = C_{0\text{-}w}G + C_{0\text{-}spm}G_{spm}$ $= Z_w f_{0\text{-}w}G + Z_{spm}f_{0\text{-}spm}G_{spm}$ $= f_{0\text{-}w}(Z_wG + 1.25Z_{spm}G_{spm})$

Initial fugacity and chemical concentration

Parameter	Symbol	Unit	Value	Reference/ Equation
Initial fugacity in water	$f_{0\text{-}w}$	Pa	2.71E-04	$f_{0\text{-}w} = N/(Z_wG + 1.25Z_{spm}G_{spm})$
Initial fugacity SPM	$f_{0\text{-}spm}$	Pa	3.43E-04	$f_{0\text{-}spm} = f_{0\text{-}w}/0.79$
Initial fugacity sediment	$f_{0\text{-}sed}$	Pa	3.43E-04	$f_{0\text{-}sed} = f_{0\text{-}w}/0.79$
Ratio of initial fugacity	$f_{0\text{-}w}/f_{0\text{-}sed}$	-	0.79	or $f_w = 0.79f_{sed}$
NP concentration in dissolved phase	$C_{0\text{-}w}$	mol/m^3	2.46E-05	$C_{0\text{-}w} = Z_w f_{0\text{-}w}$
NP concentration in SPM	$C_{0\text{-}spm}$	mol/m^3	1.02E-02	$C_{0\text{-}spm} = Z_{spm}f_{0\text{-}spm}$
NP concentration in sediment	$C_{0\text{-}sed}$	mol/m^3	1.02E-02	$C_{0\text{-}sed} = Z_{sed}f_{0\text{-}sed}$

Fugacity, concentration, and amount of NP in the evaluative environmental compartments

Parameter	Symbol	Unit	Value	Reference/ Equation
Fugacity of NP in sediment	f_{sed}	Pa	1.95E-04	$f_{sed} = N/\Sigma D$
Fugacity of NP in water	f_w	Pa	1.55E-04	$f_w = 0.79f_{sed}$
Fugacity of NP in SPM	f_{spm}	Pa	1.95E-04	$f_{spm} = f_{sed}$
Equilibrium concentration in water	C_w	mol/m^3	1.40E-05	$C_w = Z_w f_w$
	C_w	µg/L	**3.09**	$C_w(\mu g/L) = C_w(mol/m^3)M(g/mol)*10^3$
Equilibrium concentration in SPM	C_{spm}	mol/m^3	5.79E-03	$C_{spm} = Z_{spm}f_{spm}$
	C_{spm}	ng/g dw	**842**	$C_{spm} = K_{spm}C_w$

Parameter	Symbol	Unit	Value	Reference/ Equation
Equilibrium concentration in sediment	C_{sed}	mol/m^3	5.79E-03	$C_{sed} = Z_{sed}f_{sed}$
	C_{sed}	ng/g dw	**842**	$C_{sed} = K_{sed}C_w$
Amount of NP in water	m_w	mol	5.23E+02	$m_w = C_w(mol/m^3)\ V_w$
Amount of NP in SPM	m_{spm}	mol	3.07E+01	$m_{spm} = C_{spm}(mol/m^3)\ V_wSPM/10^6\rho$
Amount of NP in sediment	m_{sed}	mol	5.61E+02	$m_{spm} = C_{sed}(mol/m^3)\ V_{sed}$
Total amount	**$\sum m$**	**mol**	**1.11E+03**	

Mass-balance at equilibrium

Parameter	Symbol	Unit	Value	Reference/ Equation
Advection via bulk water	N_{Aw}	mol/h	1.87	$N_{Aw} = D_{Aw}f_w$
Advection via bulk SPM	N_{Aspm}	mol/h	0.09	$N_{Aspm} = D_{Aspm}f_w$
Diffusion between water & sediment	$N_{Dw\text{-}sed}$	mol/h	0.12	$N_{Dw\text{-}sed} = D_{Dw\text{-}sed}(f_{sed} - f_w)$
Biodegradation in bulk water	N_{Rw}	mol/h	1.08	$N_{Rw} = D_{Rw}f_w$
Biodegradation in sediment	N_{Rsed}	mol/h	0.33	$N_{Rsed} = D_{Rsed}f_{sed}$
Total amount	**N**	**mol/h**	**3.48**	

Residence time

Parameter	Symbol	Unit	Value	Reference/ Equation
Advection residence time	τ_A	day	22.3	$\tau_A = \sum m/(N_{Aw} + N_{Aspm} + N_{Dw\text{-}sed})/24$
Reaction residence time	τ_R	day	33.0	$\tau_R = \sum m/(N_{Rw} + N_{Rsed})/24$
Overall residence time	τ	day	13.3	$\tau = \sum m/N/24$

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APPENDIX 3

Contents:

Table A3.1 The riverine organisms and literature-derived characteristics

Species	Scientific name	Habitat	Feeding characteristic	Reference
Macrophytes	*Myriophyllum spicatum*	Fresh water	-	-
Phytoplankton (algae, diatom)	na	Fresh - saline water	-	-
Zooplankton (protozoa, rotifer, daphnia)	na	Fresh - saline water	Phytoplankton	Froese (2008)
Mud worm	*Perinereis nuntia*	Freshwater	Benthivore, detritivore	Kurihara (1983)
Earth worm	*Lumbriculus variegatus*	Aquatic and terrestrial	Benthivore, detritivore	Kurihara (1983)
Benthic invertebrates	(e.g. *Assiminea lutea, Corbicula lutea*)	Fresh - brackish water	Benthivore	Anh et al. (2014)
Small fish (as prey)	-	Freshwater	Phytoplankton, zooplankton, detritus	Estimated by the authors
Grass carp	*Ctenopharyngodon idella*	Freshwater	Herbivore	MDC, (2010)
Common shrimp	*Gammarus pulex*	Freshwater	Detritivore	Anh et al. (2014)
Striped/grey mullet	*Mulgil cephalus*	Fresh - saline water	Bottom dwelling omnivore (phytoplankton, zooplankton, detritus)	Islam et al. (2009)
Eel	*Monopterus albus*	Freshwater	Bottom dwelling carnivore (small fish, worm, crustacean and other small aquatic animals, detritus)	Froese (2008)
Penaied shrimp	*Metapenaeus affinis*	Fresh - brackish water	Bottom-dwelling omnivore (detritus, macrophytes, zooplankton, worms, small fish, benthic invertebrates, and shrimps)	Dincer and Aydin (2014)
Common carp	*Cyprinus carpio*	Freshwater	Omnivore (detritus, macrophytes, phytoplankton, zooplankton, ostracods, gastropods, and insects)	Dadebo et al. (2015)
Whiteleg shrimp	*Litopenaeus vannamei*	Brackish water	Omnivore (detritus, macrophytes, zooplankton, and benthos preys)	Martínez-Córdova and Peña-Messina (2005)
Crab	*Sesarma dehaani*	Freshwater	Bottom dwelling benthivore (phytoplankton, benthic micro algae, detritus is of predominant)	Tue et al. (2012)
Tilapia	*Oreochromis niloticus*	Freshwater	Planktivore (algae, diatom, zooplankton)	Getabu (1994)
Mullet	*Liza soiuy*	Fresh - saline water	Bottom dwelling omnivore (phytoplankton, plant materials, zooplankton, pisces bones and scales)	Kaniz Fatema et al. (2013)
Climbing pearch	*Anabas testudineus*	Fresh - brackish water	Omnivore (phytoplankton, zooplankton, small fish, shrimp, crustaceans, and detritus)	Roy et al. (2013)
Back seabream	*Acanthopagrus schlegelii*	Brackish water	Mollusks and polychaetes	Froese (2008)

Species	Scientific name	Habitat	Feeding characteristic	Reference
White flower croaker	*Nibea albiflora*	Brackish - saline water	Bottom dwelling carnivores (small crustaceans, fishes, and benthic organisms)	Talk-about-fish (website)
Bartail flathead	*Platycephalus indicus*	Brackish - saline water	Bottom dwelling carnivores, (small crustaceans, fishes, and benthic organisms)	Talk-about-fish (website)
Sand goby	*Glossogobius giuris*	Fresh - saline water	Bottom dwelling carnivores (zooplankton, crustaceans, small fish, and aquatic plant)	Trinh (2015)
Spotted snakehead	*Channa maculata*	Freshwater	Carnivore (crustaceans, large insects, fish and even frogs)	Froese (2008)
Walking catfish	*Clarias batrachus*	Freshwater	Omnivore (detritus, insect larvae, worms, shrimps, and small fish)	Sakhare and Chalak (2014)
Yellowfin seabream	*Sparus latus*	Brackish - saline water	Bottom-dwelling omnivore (bivalves, gastropods, shrimps, crabs, aquatic plants, and animal derivatives)	Sourinejad et al. (2015)

Table A3.2 Food web interaction among the organisms

Species	Zoo-plank ton	Mud worm	Earth worm	Benthic inverte-brates	Small fish	Grass carp	Common shrimp	Striped/ grey mullet	Eel	Penaied shrimp	Common carp	Whiteleg shrimp	Crab	Tilapia	Mullet	Climbing pearch	Back seabream	White flower croaker	Bartail flathead	Sand goby	Spotted snakehead	Walking catfish	Yellowfin seabream
Detritus		1	1	0.375	0.25		0.40	0.40	0.2	0.0119	0.398	0.7453	0.9			0.0127				0.198		0.0705	0.05
Macro-phyte					0.25	1				0.0172	0.124	0.0566			0.87					0.060			0.04
Phyto-plankton	1			0.500	0.25		0.05	0.53			0.044		0.1	0.872	0.11	0.1193							
Zoo-plankton				0.125	0.25		0.55	0.07		0.1628	0.022	0.0903		0.128	0.02	0.3138				0.060		0.2766	
Mud worm									0.1	0.2756							0.25	0.125	0.125	0.074	0.08	0.1014	
Earth worm									0.1	0.2756							0.25	0.125	0.125	0.074	0.08	0.1014	
Benthic inverte-brates									0.2	0.1336	0.048					0.2174	0.5	0.25	0.25	0.150	0.16		0.48
Small fish									0.2	0.0517								0.25	0.25	0.234	0.16	0.3027	
Grass carp																							
Common shrimp									0.2	0.0579						0.0725		0.08	0.08	0.050	0.08	0.0477	0.10
Striped/ grey mullet																							
Eel																							
Penaied shrimp																0.0725		0.08	0.08	0.050	0.08	0.0477	0.10
Common carp																							
Whiteleg shrimp																0.0725		0.08	0.08	0.050	0.08	0.0477	0.10
Crab																					0.16		0.13
Reference	Froese (2008)	Kurihara (1983)		Anh et al. (2014)	Estimat-ed by the author	MDC, (2010)	Anh et al. (2014)	Froese (2008)	Estimat-ed from Froese (2008)	Dincer and Aydin (2014)	Dadebo et al. (2015)	Martínez-Córdova and Peña-Messina (2005)	Estimated from Tue et al. (2012)	Getabu (1994)	Kaniz Fatema et al. (2013)	Roy et al. (2013)	Estimated from Froese (2008)	Estimat-ed from Talk-about-fish (website)	Estimat-ed from Talk-about-fish (website)	Trinh (2015)	Estimated from Froese (2008)	Sakhare and Chalak (2014)	Sourinejad et al. (2015)

The food web bioaccumulation model

The simplified model based on a steady-state assumption for the estimation of chemical concentration in biota is as follows (Arnot and Gobas 2004):

C_B (g/kg ww) = $[k_1(m_O\phi C_{WT,O} + m_P C_{WD,S}) + k_D\sum P_i C_{Di}]/(k_2 + k_E + k_G + k_M)$

where k_1 (L/kg.d) is the gill uptake rate constant, k_D (L/kg.d) is the dietary uptake rate constant, k_2 (d^{-1}) is the gill elimination rate constant, k_E (d^{-1}) is the fecal egestion rate constant, k_G (d^{-1}) is the growth dilution rate constant, and k_M (d^{-1}) is the metabolic transformation rate constant. k_D and k_E equal zero for both phytoplankton and macrophytes. m_O is the fraction of the respiratory ventilation that involves overlying water, m_p is the fraction of the respiratory ventilation that involves sediment-associated pore water, and ϕ is the fraction of the chemical concentration in the overlying water that is freely dissolved. $C_{WT,O}$ (g/L) is the total chemical concentration in the water column, $C_{WD,S}$ (g/L) is the freely dissolved chemical concentration in the sediment associated pore, P_i is the fraction of the diet consisting of prey item i^{th}, and $C_{D,i}$ (g/kg) is the concentration of chemical in prey item i^{th}.

The lipid weight chemical concentration:

C_B (ng/g lw) = C_B (g/kg ww)$*10^6/f_{lipid}$

The estimations of ϕ, k_1, k_2, k_D, k_E, k_G, k_M were made using sub-models as follows (Arnot and Gobas 2004).

Fraction of the chemical concentration in the overlying water that is freely dissolved:

$\phi = 1/(1 + POC.D_{POC}.\alpha_{POC}.Kow + DOC.D_{DOC}.\alpha_{DOC}.Kow)$

where *POC* and *DOC* (kg/L) are the concentrations of particulate organic carbon and dissolved organic carbon in the water, respectively; D_{POC} and D_{DOC} are the disequilibrium factors for POC-water and DOC-water partitioning, respectively; αPOC and αDOC are POC-octanol and DOC-octanol proportionality constants, respectively.

Gill uptake rate constant:

Gill uptake rate constant for zooplankton, invertebrates, and fish:

$k_1 = E_W G_V/W_B$

$E_W = 1/[1.85 + (155/K_{OW})]$

$G_V = 1400 W_B^{0.65}/C_{OX}$

where E_W is the gill chemical uptake efficiency, G_V (L/d) is the diffusion rate of the chemical across the respiratory surface area, W_B (kg) is the wet weight of the organism, K_{OW} is the octanol-water partitioning coefficient, and C_{OX} (mgO_2/L) is the dissolved oxygen concentration.

Gill uptake rate constant for phytoplankton and macrophyte:

$k_1 = 1/[A + (B/K_{OW})]$

where *A* and *B* are constants (with units of time) describing the resistance to chemical uptake through the aqueous and organic phases, respectively.

Gill elimination rate constant

$k_2 = k_1/K_{BW}$

$K_{BW} = k_1/k_2 = f_{LB}K_{OW} + f_{NB}\beta K_{OW} + f_{WB}$

where K_{BW} is the partitioning coefficient of chemical between water and the organism; f_{LB}, f_{NB}, and f_{WB} indicate lipid, non-lipid, and water fractions, respectively, in biota; β is the non-lipid organic matter-octanol proportionality constant.

Dietary uptake rate constant (for zooplankton, invertebrates, and fish)

$k_D = E_DG_D/W_B$

$G_D = 0.022W_B^{0.85}\exp(0.06T)$ (T = 2 - 25°C)

where E_D is the dietary chemical transfer efficiency (E_D = 0.5 for continuously fed invertebrates and fish), and G_D (kg/d) is the feeding rate.

The fecal egestion rate

$k_E = E_DG_FK_{GB}/W_B$

$G_F = [(1-\varepsilon_L)f_{LD} + (1-\varepsilon_N)f_{ND} + (1-\varepsilon_W)f_{WD}]G_D$

$K_{GB} = (f_{LG}K_{OW} + f_{NG}\beta K_{OW} + f_{WG})/(f_{LB}K_{OW} + f_{NB}\beta K_{OW} + f_{WB})$

$f_{LG} = (1 - \varepsilon_L)f_{LD}/[(1 - \varepsilon_L)f_{LD} + (1 - \varepsilon_N)f_{ND} + (1 - \varepsilon_W)f_{WD}]$

$f_{NG} = (1 - \varepsilon_N)f_{ND}/[(1 - \varepsilon_L)f_{LD} + (1 - \varepsilon_N)f_{ND} + (1 - \varepsilon_W)f_{WD}]$

$f_{WG} = (1 - \varepsilon_W)f_{WD}/[(1 - \varepsilon_L)f_{LD} + (1 - \varepsilon_N)f_{ND} + (1 - \varepsilon_W)f_{WD}]$

Where G_F (d^{-1}) is the fecal egestion rate, and K_{GB} is the partition coefficient of the chemical between the gastro-intestinal tract (GIT) and the organism. ε_L, ε_N, and ε_W are the dietary absorption efficiencies of lipid, NLOM, and water, respectively; f_{LD}, f_{ND}, and f_{WD} are the lipid, NLOM, and water contents, respectively, in the diet; f_{LG}, f_{NG}, and f_{WG} are the lipid, NLOM, and water contents, respectively, in the gut.

Growth dilution rate constant

$k_G = 0.00251W_B^{-0.2}$ at T = 25°C

Metabolic transformation rate constant for NP

The biotransformation rate for fish at the original mass and temperature *($k_{M,i}$)* was derived by conversing the biotransformation rate normalized to a mass (10 g) and temperature (15°C) *($k_{M,N}$)* using the following equation (Arnot et al. 2008):

$k_{M,i} = k_{M,N}/[(0.01/W_{Bi})^{-0.25}\exp[0.01(15 - T_i)]$

where the average $k_{M,N}$ value for NP among fish is 1.7 ± 3.9 (d^{-1}).

Bioaccumulation and biomagnification factors

Bioaccumulation and biomagnification factors for the organisms in the riverine food web were calculated using the following equations (Arnot and Gobas 2006):

BAF_ww = C_B (g/kg ww)/$C_{WT,O}$

BAF_lw = C_B (ng/g lw)*10^{-6}/$C_{WT,O}$

BMF_lw = C_B (ng/g lw)/C_D (ng/g lw) *or/and* (ng/g OC)

where BAF_ww and BAF_lw (L/kg) are bioaccumulation factors on wet weight and lipid weight basis, respectively; $C_{WT,O}$ (g/L) denotes the total chemical concentration in the bulk water phase; BMF_lw indicates lipid-normalized biomagnification factor.

Table A3.3 Parameters for estimating rate constants and bioaccumulation factors with the food web model

Symbol	Definition	Unit	Value	Reference/Equation
$C_{WT,O}$	The total chemical concentration in the water column above the sediment	g/L	Low scenario: 5.09E-06 Medium scenario: 4.30E-06 High scenario: 3.24E-06	$C_{WT,O} = [C_W(\mu g/L) + C_{spm}(ng/g\ dw)*SPM(mg/L)*10^{-6}]10^{-6}$ C_W and C_{spm} are outcomes of the fugacity-based fate model
C_{sed}	The equilibrium chemical concentration in the sediment	g/kg dw	Low scenario: 1.32E-03 Medium scenario: 1.12E-03 High scenario: 8.42E-04	Outcome of the fugacity-based fate model
POC	Particulate organic carbon content	kg/L	6.89E-06	$POC(kg/L) = POC(\%)*SPM(mg/L)*10^{-6}$
DOC	Particulate organic carbon content	kg/L	1.09E-05	Estimated using the equation suggested by Gong et al. (2012): $C_w(ng/L) = 359.7DOC(mg/L) + 176.5$ ($R = 0.74$, $p < 0.0005$)
D_{POC}	The disequilibrium factor for POC-water partitioning	-	1	It was assumed that POC-water and DOC-water partitioning were at equilibrium.
D_{DOC}	The disequilibrium factor for DOC-water partitioning	-	1	
α_{POC}	POC-octanol proportionality constant	-	0.35	Seth et al. (1999)
α_{DOC}	DOC-octanol proportionality constant	-	0.08	Burkhard (2000)
K_{OW}	Octanol-water partitioning coefficient	L/kg	30,199.5	Ahel and Giger (1993)
C_{OX}	Dissolved oxygen concentration	mgO_2/L	2.24	SIFEB (2013)
A	Resistance to chemical uptake through the aqueous phase constant	d	6.0E-05	Swackhamer and Skoglund (1993), Gobas and MacLean (2003)
B	Resistance to chemical uptake through the organic phase constant	d	5.5	Wang et al. (1996), Koelmans et al. (1999), Koelmans et al. (1995), Koelmans et al. (1993)
f_{LB}	Lipid fraction of biota	kg/kg		$f_{LB} = 0.002$ for detritus (Kuchkina et al. 2011) Lipid fraction of biota is presented in Table 4.3
f_{NB}	Non-lipid fraction in biota	kg/kg	Phytoplankton & macrophytes: $f_{NB} = 0.01$ Other species: $f_{NB} = 0.2$	Arnot and Gobas (2003)
f_{WB}	Water fraction in biota	kg/kg		$f_{WB} = 1 - (f_{LB} + f_{NB})$
β	Sorption capacity of non-lipid organic matter (NLOM) to that of octanol	-	Phytoplankton & macrophytes: $\beta = 0.35$ Other species: $\beta = 0.035$	Arnot and Gobas (2004)
m_O	The fraction of the respiratory ventilation that involves overlying water	%	$m_O = 100 - m_P$ Benthivores: $m_O = 95$ Non-benthivores: $m_O = 100$	Arnot and Gobas (2004)

Symbol	Definition	Unit	Value	Reference/Equation
m_P	The fraction of the respiratory ventilation that involves sediment-associated pore water	%	Benthivores: $m_P = 5$ Non-benthivore: $m_P = 0$	Arnot and Gobas (2004)
ε_L	The dietary assimilation efficiencies of lipid ε_L	%	Zooplankton: $\varepsilon_L = 72$ Invertebrates: $\varepsilon_L = 75$ Fish: $\varepsilon_L = 92$	ε_L and ε_N for zooplankton (Arnot and Gobas 2004) ε_L and ε_N for invertebrates (Arnot and Gobas 2004) ε_L and ε_N for fish (Gobas et al. 1999, Nichols et al. 2001)
ε_N	The dietary assimilation efficiencies of NLOM	%	Zooplankton: $\varepsilon_N = 72$ Invertebrates: $\varepsilon_N = 75$ Fish: $\varepsilon_N = 60$	
ε_W	The dietary assimilation efficiencies water	%	All species: $\varepsilon_W = 0.25$	Arnot and Gobas (2004)
f_{LD}	Overall lipid content of the diet	kg/kg		estimated from the food web interaction (Table A3.2)
f_{ND}	Overall NLOM content of the diet	kg/kg		
f_{WD}	Overall water content of the diet	kg/kg		
P_i	The fraction of the diet consisting of prey item i	%		Table A3.2

Table A3.4 Chemical uptake and elimination rate constants for fish

Species	Scientific name	k_1 (L/kg.d)	k_2 (d^{-1})	k_D (L/kg.d)	K_E (d^{-1})	K_G (d^{-1})	K_M (d^{-1})
Small fish (as prey)	-	2.152E+03	1.719	0.109	0.010	0.007	2.234
Grass carp	*Ctenopharyngodon idella*	3.369E+02	0.495	0.049	0.001	0.003	0.594
Striped/grey mullet	*Mulgil cephalus*	4.732E+02	0.271	0.057	0.003	0.003	0.757
Eel	*Monopterus albus*	5.211E+02	1.266	0.059	0.029	0.003	0.811
Common carp	*Cyprinus carpio*	4.966E+02	0.565	0.058	0.007	0.003	0.784
Tilapia	*Oreochromis niloticus*	4.966E+02	0.444	0.058	0.004	0.003	0.784
Mullet	*Liza soiuy*	7.736E+02	0.506	0.070	0.001	0.004	1.076
Climbing pearch	*Anabas testudineus*	8.545E+02	1.495	0.073	0.012	0.004	1.155
Back seabream	*Acanthopagrus schlegelii*	5.135E+02	0.229	0.059	0.006	0.003	0.803
White flower croaker	*Nibea albiflora*	5.970E+02	0.514	0.063	0.010	0.003	0.894
Bartail flathead	*Platycephalus indicus*	4.831E+02	0.493	0.057	0.011	0.003	0.769
Sand goby	*Glossogobius giuris*	7.388E+02	1.195	0.069	0.020	0.004	1.041
Spotted snakehead	*Channa maculata*	2.923E+02	0.385	0.046	0.010	0.002	0.537
Walking catfish	*Clarias batrachus*	4.966E+02	0.382	0.058	0.009	0.003	0.784
Yellowfin seabream	*Sparus latus*	5.048E+02	0.188	0.058	0.002	0.003	0.793

Table A3.5 Chemical uptake and elimination rate constants for other species

Species	Scientific name	k_1 (L/kg.d)	k_2 (d^{-1})	k_D (L/kg.d)	K_E (d^{-1})	K_G (d^{-1})	K_M (d^{-1})	Reference for k_M estimation
Macrophytes	*Myriophyllum spicatum*	4.130E+03	24.719	-	-	Mean k_G (d^{-1}) = 0.08[a]	0.1[b]	[a] Swackhamer and Skoglund (1993) [b] based on algae (*Cyclotella caspia*) (Liu et al. 2013)
Phytoplankton (algae, diatom)	na	4.130E+03	5.254	-	-			
Zooplankton (protozoa, rotifer, daphnia)	na	1.156E+05	150.547	0.599	0.151	0.071	10.2	based on water fleas (*Daphnia magna*) (Preuss et al. 2008)
Benthic invertebrates	(e.g. *Assiminea lutea, Corbicula lutea*)	2.152E+03	8.559	0.109	0.075	0.007		
Mud worm	*Perinereis nuntia*	4.283E+03	1.006	0.146	0.006	0.011	0.235	estimated from the difference between the elimination rates of 0.251 d^{-1} (Nurulnadia et al. 2013) and the predicted egestion rate constants and growth dilution rate constants
Earth worm	*Oligochaeta (Lumbriculus variegatus)*	2.265E+04	5.837	0.298	0.013	0.028	0.151	estimated from the difference between the elimination rates of 0.192 d^{-1} (Maenpaa and Kukkonen 2006) and the predicted egestion rate constants and growth dilution rate constants
Common shrimp	*Gammarus pulex*	9.613E+02	1.514	0.077	0.022	0.005	0.214	estimated from the difference between the elimination rates of G. pulex of 0.24 d^{-1} (Gross-Sorokin et al. 2003) and the predicted egestion rate constants and growth dilution rate constants
Penaied shrimp	*Metapenaeus affinis*	1.465E+03	2.423	0.092	0.096	0.006	0.138	
Whiteleg shrimp	*Litopenaeus vannamei*	1.442E+03	0.837	0.092	0.008	0.006	0.227	
Crab	*Sesarma dehaani*	1.337E+03	1.626	0.089	0.018	0.006	0.217	

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APPENDIX 4

Contents:

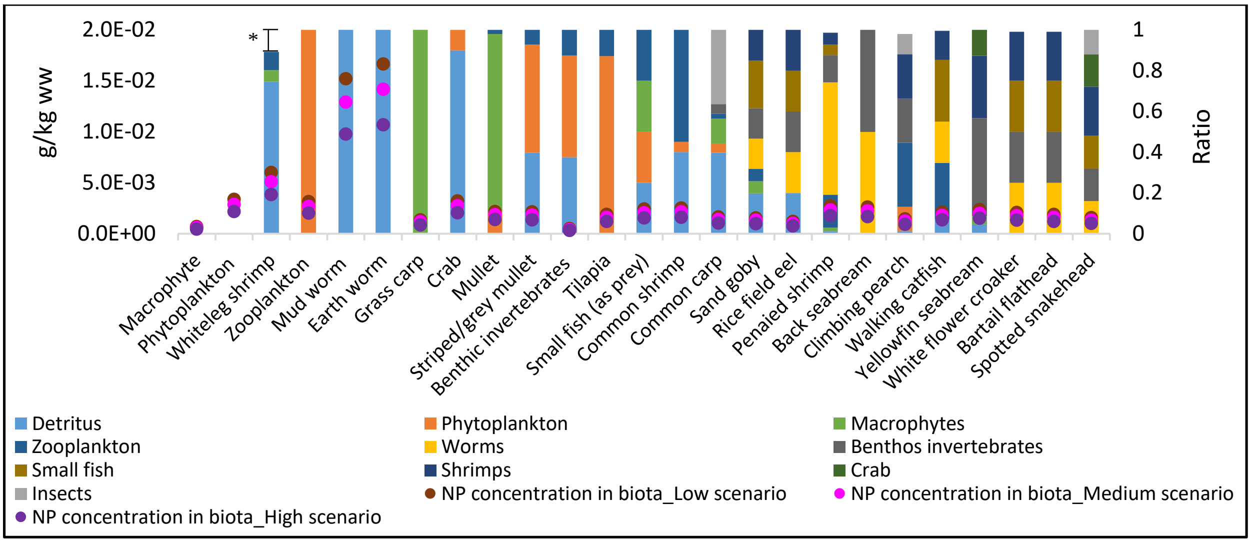

Figure A4.1 Diet compositions in relation to wet weight NP concentrations of the biota under three scenarios

Notes: * unidentified benthos preys. Macrophyte and phytoplankton: chemical uptake occurs via the aqueous and organic phases only.

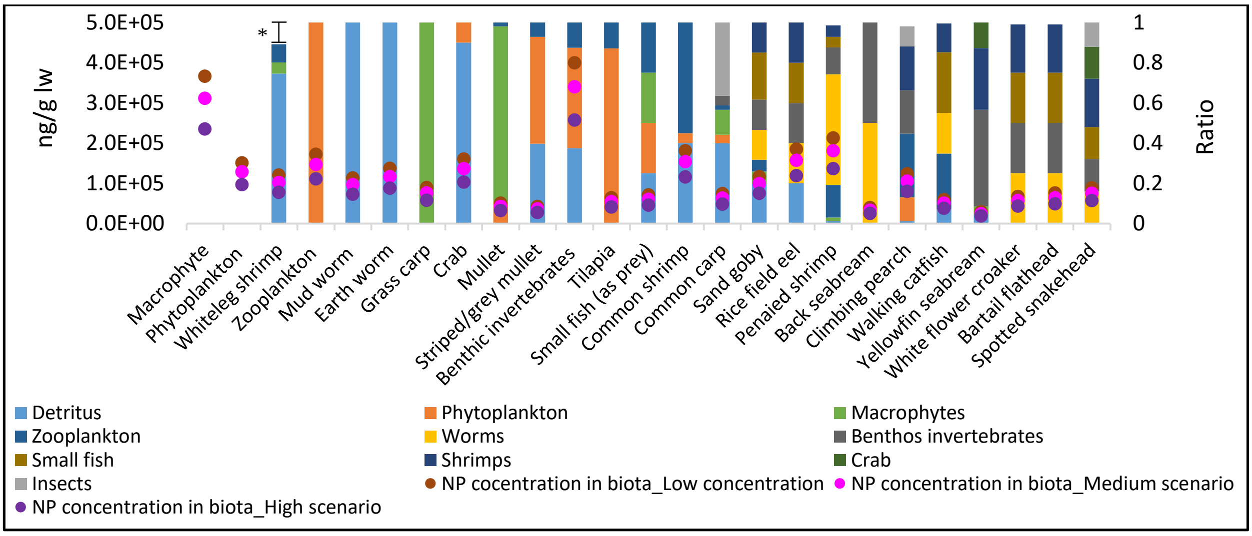

Figure A4.2 Diet compositions in relation to lipid-normalized NP concentrations of the biota under three scenarios

Notes: * unidentified benthos preys. Macrophyte and phytoplankton: chemical uptake occurs via the aqueous and organic phases only.

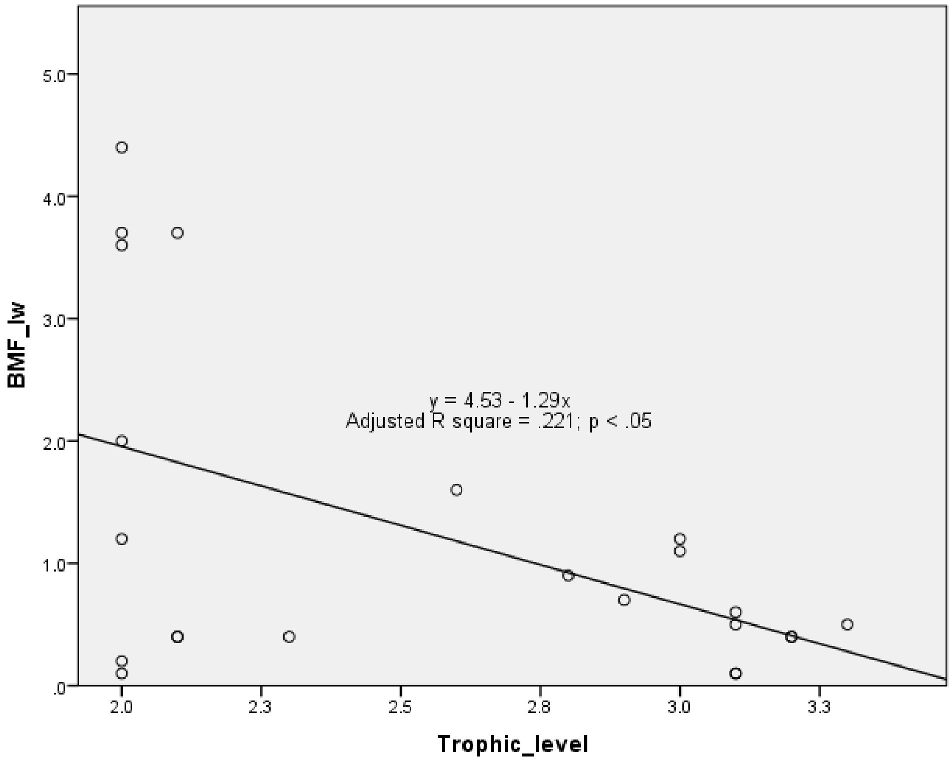

Figure A4.3 Biomagnification factors versus trophic levels under medium scenario

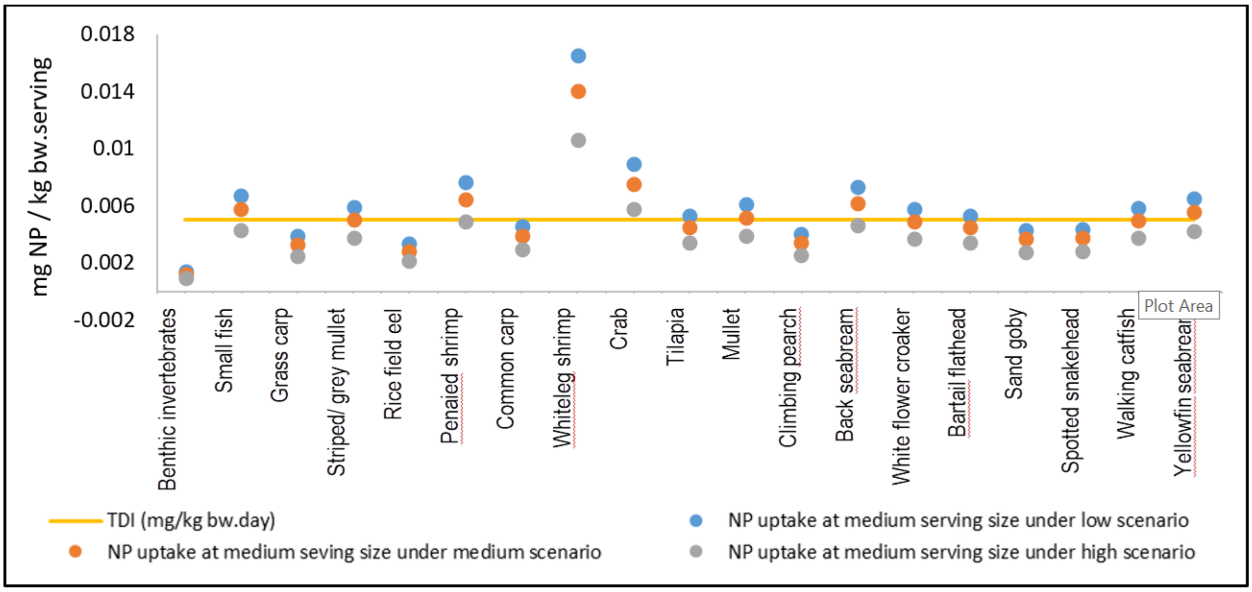

Figure A4.4 NP intake from consuming each riverine food product

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