



Development of human health risk-based Soil Quality Standards for Turkey: Conceptual framework

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ABSTRACT

Soil Quality Standards (SQSs) are considered as an important tool for management of contaminated sites. In order to manage adverse effects associated with soil contamination, risk-based SQSs are used worldwide. However, developing health risk-based SQSs is a challenging task. It involves comprehensive assessment of nationwide site characteristics (e.g. soil, hydrogeological, climatic properties), understanding of social behaviors that determine human exposure to soil pollutants (e.g. exposure scenarios and parameters), and making political decisions with regard to the economical concerns. To set fixed levels of SQSs that apply for various cases and that is protective of human health, conservative assumptions should be made. The level of conservativeness of the standards should not pose a challenge to the objectivity of the decision given and not force technical and financial resources. Regarding this challenge, a novel conceptual framework is developed for derivation of human health risk-based SQSs for Turkey. In this paper, the main elements of the established conceptual framework, the methods used for specification of the generic site characteristics, the information sources used for compilation of needed chemical and toxicological data, the political decisions taken, and the challenges encountered during these studies are presented. It is believed that the presented road map developed through the conceptual framework will be beneficial for other countries that are in the stage of deriving SQSs.

1. Introduction

Together with air and water, soil constitutes the environment to sustain the life on earth. These three components are intimately related to each other through *hydrologic cycle*, which defines the occurrence, exchange and movement of water on the earth. In this regard, soil has two fundamental environmental functions; soil acts as (i) a *receiving medium* for disposal of wastes and (ii) a *natural recharge zone* for surface water and groundwater resources (Ünlü et al., 2009). As a consequence, the quantity and the quality of surface water and groundwater are directly affected by soil, and because of its role in hydrologic cycle, soil appears as a key element in terms of environmental pollution problems. Since soil serves as an underlying material for various human activities, it became a receiving body for various contaminants. In many countries, poor waste management practices, discharge of wastewater, excessive and uncontrolled use of herbicides and pesticides for agricultural purposes, spills, leaks and/or discharge of chemicals during handling, storage and transportation result in complex environmental problems threatening the human health and the ecology (Swartjes, 2011). Because of the widespread occurrence of soil contamination events and its effects on human health and environment, soil quality has been an issue taken more seriously during the last three decades. Over this period,

significant improvements have been recorded related to identification, registration, assessment, and clean-up of contaminated sites in terms of policy framework and technological developments. Since the number of contaminated sites increased significantly, mostly due to industrial activities, many countries, environmental agencies and organizations configuring their policy framework by integrating efficient management systems for contaminated sites (Rodrigues et al., 2009a)

In this respect, most EU Member States and the North American Countries define risk-based approaches as the best available strategy for dealing with the problems posed by soil contamination, assessing the need for cleanup and planning remedial actions. Because of the success achieved, the risk-based site management approaches have been included in the environmental policy of many countries (NICOLE, 2002), and risk-based SQSs are used in most European Countries (e.g., Austria, Belgium, Germany, the Netherlands, United Kingdom, Norway, Sweden, Spain, etc.), Canada and the USA either to define target levels, or to screen sites that do not need further investigation, or to determine the urgency of remediation, or as clean-up levels (Rodrigues et al., 2009a).

FAO (2018) recently identified the soil pollution as one of the main threats affecting global soils and the ecosystem services provided by soils; and hence, urged that national governments implement SQSs in regulating soil pollution and remediating contaminated soils to protect

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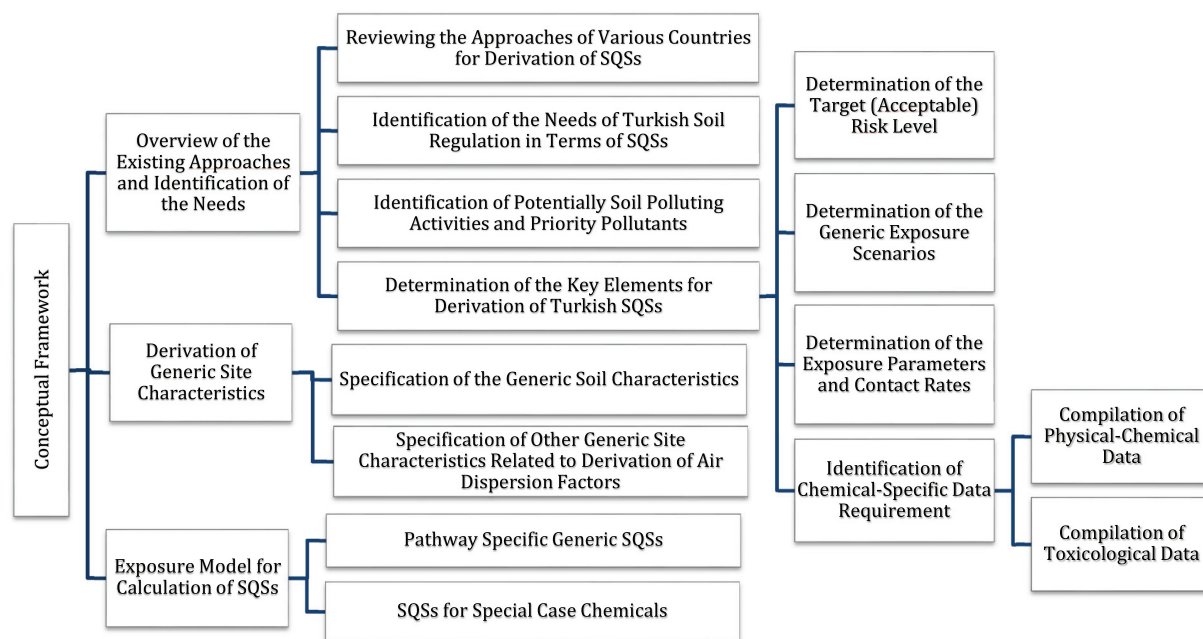


Fig. 1. Conceptual framework for developing health-risk-based Soil Quality Standards (SQSs) for Turkey.

the human health and the environment. As an important tool of contaminated site management, the intended purpose of use of the risk-based SQSs varies; and thus they have to be developed based on the specific needs of the country. Because derivation of risk-based SQSs, which indeed is a highly challenging process, requires a comprehensive assessment and understanding of the environmental, socio-cultural, and economical conditions of the country. Some recent examples on the development and implementation of national soil policy framework are presented by Rodrigues et al. (2009a, 2009b) for Portugal, by Swartjes et al. (2012) and de Snoo (2020) for Netherlands, and by the Finnish Ministry of Environment (FME (Finnish Ministry of the Environment), 2017) for Finland. Along the same line, Kentel et al. (2011) presented, from a broader perspective, the challenges of implementation of a newly developed risk-based contaminated site management system in Turkey as whole, of which the derivation of SQS is certainly an important component.

While developing country-specific SQSs, the regulatory needs of the country in terms of management of contaminated sites and priority pollutants based on the common potentially polluting activities taking place within the country should be identified, and the key elements for derivation of SQSs should be specified. Risk-based SQSs are conceptually rely on the source-pathway-receptor analysis involving assessment of source characteristics (e.g., source type, size and depth, contaminants of concern and their fate and transport characteristics), site characteristics (e.g., soil, geology, hydrology, climatic conditions, etc.), receptor characteristics (e.g., exposure duration and frequency, body weight, etc.), and the potential exposure pathways (e.g., ingestion, dermal contact, inhalation, etc.) that connect the source to the receptors. And finally, it requires determination of generic numerical values of risk assessment parameters that are reasonable to apply for various contaminated site cases, conservative enough to protect human health, and objective not to force technical and financial resources.

Considering all possible challenges, a conceptual framework that can serve as a well-defined road map for the entire process of deriving country specific risk-based SQSs has been lacking. This paper attempts to fill this gap by proposing a novel conceptual framework through a holistic approach for the derivation of human health risk-based SQSs for Turkey. In an earlier paper, Kentel et al. (2011) emphasized challenges regarding institutional and technical capacity building needs to carry

out risk-based site assessment and remedial investigation activities effectively rather than derivation of SQSs per se. This new manuscript, however, specifically focuses on developing a novel conceptual framework for derivation of pathway and chemical specific SQSs as well as on the related data needs, tolls and all other technical details, as described in detail throughout the manuscript. In this regard, this paper complements the earlier paper by narrowing down to derivation of SQSs only in a wide range of issues related to effective implementation of risk-based contaminated site management system.

In the following, we present the main elements and the essential steps of the established conceptual framework, the methods used for specification of generic characteristics, the information sources used for compilation of required chemical and toxicological data, the political decisions taken, and the challenges encountered during these studies. Having known the urgency pointed out by the FAO (2018), the presented road map through the developed conceptual framework is of importance for other countries that are in the planning stage of deriving SQSs.

2. Conceptual framework

Although development of risk-based SQSs relies on the same principles, different countries end up with different generic standards depending on the scientific approaches they adopt, the use of different assumptions, toxicological data, generic site-characteristics, accepted risk level, etc. (Carlon et al., 2007). Therefore, it was essential to review the approaches used in different countries to attain a background and to establish a conceptual framework for the development of SQSs.

The conceptual framework established to develop generic SQSs for Turkey involved accomplishment of three main tasks (Fig. 1). The first task was to have in depth overview of the existing approaches used for derivation of health-risk-based SQSs; and through this, identification of the information needs. The second task was the derivation of the generic site characteristics for Turkey to be used in the development of risk-based SQSs. In this respect, the specification of the generic soil characteristics, derivation of air dispersion factors that represent the dispersion of soil contaminant (i.e., volatiles and fugitive dusts) emissions in air under regional meteorological conditions, and specification of the other relevant site characteristics that control generation of fugitive dust emissions were needed. The third and final task was development of a

computational procedure (called exposure model), which includes the physical-chemical and toxicological data libraries embedded in its structure, to facilitate calculation of generic and site-specific SQSs and to eliminate the potential calculation mistakes that may result due to large number of parameter values and chemical-specific data used in calculations. Consequently, availability of such a computational tool used for derivation of human health risk based SQSs ensures the sustainability, maintenance and dynamism of the Turkish contaminated sites management system.

The above-mentioned essential steps of the conceptual framework for derivation of health-risk-based SQSs for Turkey are schematically illustrated in Fig. 1. Each of these steps are discussed in the following sections.

2.1. Overview of the existing approaches and identification of the needs

In the first task, the approaches of European and North American Countries for development of health-risk-based SQSs were investigated thoroughly; the scientific approaches, the assumptions made, and the data used in derivation of standards were compared. The regulatory needs of Turkey in terms of contaminated site management were evaluated and the role of SQSs in the management system was specified. In order to determine the priority pollutants to be included in the regulation, potential soil polluting activities in Turkey and potential soil pollutants arising from these activities were identified. With the insight gained from other countries' approaches, the most appropriate approach for development of Turkish SQSs and the necessary modifications were identified taking into account of the regulatory needs. The key elements for derivation of health-risk-based SQSs, such as acceptable (target) risk level, generic exposure (land use) scenarios, relevant exposure pathways, generic values for the exposure parameters and contact rates, generic soil characteristics and chemical-specific data requirement were determined.

2.1.1. Reviewing the approaches of various countries for derivation of SQSs

The approaches of other countries to development of SQSs and the way of implementing their SQSs in the regulatory framework were reviewed. Due to their experiences in contaminated site assessment and the availability of detailed documented information, the SQSs of the USA, Germany, Netherlands, Canada and Norway were studied in detail. In addition to these countries, the similarities and distinctions between the approaches of several other countries (e.g., Austria, Belgium, Czech Republic, Denmark, Finland, France, Italy, Lithuania, Poland, Slovak Republic, Spain, Sweden, and the United Kingdom) for development of SQSs were reviewed and summarized as given in Table 1 to ease the comparison.

As seen from Table 1, in most of the European countries, the USA and Canada, traditional concentration based soil quality criteria are replaced by risk based soil quality criteria. Although the basic principles for derivation of risk-based soil quality criteria are more or less the same, different approaches and assumptions were adopted by different countries. For example; in some of the countries, only human health risks are considered for development of soil quality criteria (e.g., the USA, France), whereas ecological risks are considered in addition to health-risks in some other countries (e.g., Austria, Belgium, Canada, Finland, Germany, the Netherlands, Norway, Spain, etc.).

Some countries (e.g., Denmark, Netherlands, Norway, Slovak Republic) use single *generic* soil quality criteria that are not based on a specific land use or pathway, while some of the countries, such as Austria, Belgium, Canada, Czech Republic, etc. use soil quality criteria that are defined on the basis of different land use types, or sensitivity of the site (e.g., France, Sweden). Different from these approaches, the USA provide soil criteria based on the *generic human exposure pathways*.

Although the principle underlying the development of health-risk-based soil quality criteria is common for all countries, use of different assumptions and different parameter values in calculations result

in 10 to 100 folds of difference in SQSs for the same contaminant. One of the most important parameter that affects the value of soil quality criteria profoundly is the *acceptable risk* determined for non-threshold compounds, which varies from 10^{-4} to 10^{-6} in different countries. In most of the countries, acceptable risk is determined as 10^{-5} (e.g., Belgium, Denmark, Finland, Germany, Norway, Sweden, Italy) whereas the Netherlands accepts 10^{-4} , and the US and Canada accept 10^{-6} risk level (Carlon et al., 2007). Additionally, the differences in soil quality criteria are also attributable to the variability in environmental, socio-cultural, regulatory, political, and scientific concerns (Carlon et al., 2007). The latter item involves consideration of different exposure scenarios (different land use types and exposure pathways), use of different values for variables (exposure duration and frequency, body weight, soil ingestion amount, inhalation rate, dilution factor etc.), use of different toxicological data, and consideration of different risk levels (10^{-4} – 10^{-6}).

As much as the approach and the assumptions adopted for development of the soil quality criteria, the purpose of use of these standards (in other words, the role of soil quality criteria), which varies from country to country, is also important. Carlon et al. (2007) classifies soil quality criteria based on their role in contaminated sites management system as: *negligible risk*, *intermediate (warning) risk*, and *potentially unacceptable risk*. The soil quality criteria that are considered as long-term target values are classified within *negligible risk values* and they are mostly not related to the land use type. In most of the countries, the soil quality criteria, which designate *intermediate risk*, require further investigation in case it is exceeded. On the other hand, soil quality criteria that take place within *potentially unacceptable risk* indicate the need for remediation. The two levels of criteria, target value and intervention value, proposed by the Netherlands have different purpose of use. Target value is considered as the long-term objective and the site concentrations below this value pose *negligible risk*, meaning that the site requires no further action. However, intervention value designates *unacceptable risk* and the site concentrations exceeding this limit requires further investigation (Swartjes et al., 2012).

2.1.2. Identification of the needs of Turkish soil regulation in terms of SQSs

Developing SQSs mainly requires understanding the regulatory needs and identification of the potential difficulties in implementation of the standards, which is mostly correlated with economical concerns and technical infrastructure. For this reason, the historical evolution of Turkish regulations on soil protection were briefly summarized below from early 2000s to present. The regulatory needs in terms of contaminated site management were evaluated and the role of the SQSs in the management system was specified.

In Turkey, soil pollution due to improper waste management activities has been the main concern over the last two decades. Soil pollution problems initially were covered by the Regulations on Solid Waste Control and Hazardous Waste Control, published first in 1995 by Turkish Ministry of the Environment and Urban Affairs (TMOEUA). As a result of increasing attention to soil pollution, the Soil Pollution Control Regulation (SPCR) was published in 2001. The regulation was focusing on mainly the needs for application of stabilized sludge and composts on soils, and it was including some pre-specified fixed limiting values for a few number of inorganic and organic substances. According to the regulation, soils satisfying these standards were accepted as clean. Otherwise, any contamination had to be cleaned up to these levels. In 2005, the regulation was updated by some administrative arrangements (TMOEUA, 2005). The regulation was indicating the importance of protecting soil against pollution, and the need for confinement, investigation, monitoring and remediation of soils contaminated due to industrial activities and accidents; however, it was still lacking the insight of the latest risk-based approaches for contaminated site management. A systematic and holistic approach to be followed or the procedures and methods to be applied for identification, registration, assessment, clas-

Table 1Comparison of SQSs of different countries (adopted from [Carlon et al., 2007](#)).

	Land Use	Pathways		Soil criteria	Risk level	Purpose of use
Canada	- agricultural - residential/park lands - commercial - industrial	- soil ingestion - dermal contact - inhalation of dust - ingestion of groundwater	- inhalation of vapors (indoor air) - consumption of homegrown products - consumption of meat - consumption of dairy	Soil Quality Guidelines	Negligible Risk	Designate clean up levels
Germany	- agricultural - green land - parks/recreation - playground - residential - industrial	- soil ingestion - dust ingestion - dermal exposure	- ingestion of soil attached to homegrown vegetables - consumption of homegrown fruits	Trigger Values	Intermediate Risk	Define the need for further investigation
		- inhalation of vapors (outdoor) - ingestion of soil attached to homegrown fruits	- ingestion of soil attached to homegrown fruits - consumption of groundwater	Action Values	Unacceptable Risk	Determine the presence of soil degradation
		- inhalation of dust (outdoor) - consumption of homegrown vegetables	- consumption of groundwater	Site-Specific Intervention Values	Unacceptable Risk	Define the need for remediation and target
Netherlands	- generic	- soil ingestion - dust ingestion - dermal exposure	- inhalation of groundwater vapors - consumption of homegrown vegetables	Target Value	Negligible Risk	Long term objectives
		- inhalation of vapors (outdoor) - inhalation of dust (outdoor)	- ingestion of soil attached to homegrown vegetables - consumption of groundwater	Intervention Value	Unacceptable Risk	Define need for site-specific assessment
		- dermal exposure to dust (indoor) - inhalation of vapors (indoor)	- inhalation of volatilized domestic water - showering (dermal contact and inhalation)	Site-Specific Risk Assessment	Unacceptable Risk	Define urgency of remediation. Remedial concentration targets for immobile contaminants
		- consumption of drinking water - consumption of vegetables - consumption of fish				Further investigation
Norway	- generic	- soil ingestion - dermal contact - inhalation of dust (outdoor) - inhalation of vapors (indoor)	- consumption of drinking water - consumption of vegetables - consumption of fish	Soil Quality Guidelines	Intermediate Risk	
Sweden	- sensitive land uses - less sensitive with or without groundwater protection	- soil ingestion - dust ingestion - dermal exposure	- inhalation of vapors (indoor) - consumption of homegrown vegetables	Trigger values	Intermediate Risk	Further investigation
		- inhalation of dust (outdoor) - dermal exposure to dust (indoor)	- consumption of homegrown fruits - consumption of groundwater - consumption of fish and shell-fish			
United Kingdom	- allotments - natural - residential with plant uptake - residential without plant uptake - commercial/industrial	- soil ingestion - dust ingestion - dermal exposure	- dermal exposure to dust (indoor) - inhalation of vapors (indoor)	Soil Screening Values (SSVs) for ecological receptors	Intermediate Risk	Triggers further investigation
		- inhalation of vapors (outdoor) - inhalation of dust (outdoor)	- consumption of homegrown vegetables - ingestion of soil attached to homegrown vegetables	Refinements made to measured field concentrations and then compared with SSVs for ecological receptors	Intermediate Risk	Triggers for further investigation
				Soil Guideline Values	Unacceptable Risk	Site specific assessment required
				Site-Specific Intervention Values	Unacceptable Risk	Define the need and target for remediation
US EPA	- residential - commercial/industrial: outdoor worker - commercial/industrial: indoor worker	- ingestion - dermal absorption - inhalation of fugitive dusts (outdoor)	inhalation of vapors (outdoor) migration to groundwater	Soil Screening Levels	Intermediate Risk	Define the need for action, or further investigation

sification and remediation of contaminated sites were not defined in the regulation.

However, Turkey was experiencing some serious pollution incidences pressing the need for a comprehensive soil pollution regulation. These environmental pollution problems were mainly due to industrial accidents, chemical spills and leakages from storage tanks, illegal waste dumping, and spills from petroleum transfer lines. Therefore, a comprehensive soil regulation, which defines the state-of-art for management of contaminated sites and relevant SQSs to be used as a tool for assessment of soil quality, were required to fulfill the needs of TMOEUA. The new regulation on Soil Pollution Control and Sites Contaminated by Point Sources, which includes a comprehensive contaminated sites management system and human health risk based SQSs, was published by the TMOEUA and became legally active as of 8 June 2010.

2.1.3. Identification of potentially soil polluting activities and priority pollutants

SQSs are regarded as fundamental criteria in determination of soil pollution. It is important to develop SQSs for as many important chemicals as possible to ease decision making in case of a soil pollution event. On that account, it is essential to identify the common potentially soil polluting activities in the country and the potential soil pollutants arising from these activities in order to form the list of priority pollutants for which SQSs should be derived and included in the regulation.

Ferguson (1999) emphasizes that soil contamination events are mostly associated with industrial activities. This fact is also supported by the declaration of European Environment Agency (EEA) that “about 41% of the soil contamination is resulting from industrial and commercial services” (URL 1, 2011). In this regard, many industrial activities are considered to be the main cause of soil pollution and the sites over which such activities take place are considered as potentially contaminated sites. However, a site can be defined as contaminated, if it is polluted by *anthropogenic dangerous substances posing significant risk to human health or the environment* (EU COM, 2006a). Therefore, for identification of potentially polluting activities and the contaminants associated with those industries, it is important to identify the industries that include/generate hazardous substances in their processes. Although the Regulation on Soil Pollution Control and Sites Contaminated by Point Sources (TMOEUA, 2010) stipulates the registration of the contaminated sites in the country, a complete inventory of contaminated sites does not exist yet. Thus, satisfactory information on the common pollutants observed at the contaminated sites in Turkey could not be obtained. However, it is known that mining activities, wood processing, petroleum refining, natural gas purification and pyrolytic treatment of coal, inorganic and organic chemical processes, thermal processes, chemical surface treatment and coating of metals and other materials, non-ferrous hydrometallurgy, oily wastes and wastes of liquid fuels, organic solvents, refrigerants and propellants, electrical and electronic equipment, batteries and accumulators, wastes from human or animal health care and/or related research, wastes from incineration or pyrolysis of waste, and municipal wastes are among the activities in Turkey that include/generate significant amounts of hazardous substances in their processes (Yilmaz, 2006). Among these activities, organic chemical industry is estimated to have the highest portion in hazardous waste generation being about 480,000 tons/yr (Yilmaz and Yetiş, 2009). The sub-groups of this sector that generate considerable amounts of hazardous wastes are “pharmaceutical industry (30%), organic plant production and biocides (26%), plastic manufacture (21%), organic dye and pigment (20%) and fats, grease, soaps manufacture (3%)” (Yilmaz and Yetiş, 2009).

In this context, the potentially soil polluting activities and the most frequently observed soil pollutants in the European Countries were also reviewed. The primary sources of soil pollution in Europe are presented by European Environment Agency (EEA) (URL 1, 2011) with respect to percent of the number of sites where preliminary investigations have

been completed. According to this, industrial production and commercial services, municipal waste treatment and disposal, industrial waste treatment and disposal are shown among the foremost activities causing soil pollution. Handling losses, leakages from tanks and pipelines, and accidental spills of chemicals at industrial and commercial sites are reported to be the main sources of soil pollution in Europe (URL 1, 2011). Especially, the chemical and metal working industries, energy production and the oil industry are considered among important sources of soil pollution, while the gasoline stations and dry cleaners are regarded as the sources that pose significant environmental and human health effects in some of the European countries.

In fact, the profile of common industrial activities in Turkey shows that the potentially soil polluting activities in Turkey are comparable with the ones experienced in the European countries. Although the main sources of soil pollution and the main pollutants vary from country to country, heavy metals and mineral oil are found to be the most frequent soil pollutants in Europe, whereas mineral oil and chlorinated hydrocarbons are found to be the most frequent pollutants in groundwater (URL 1, 2011). Polycyclic aromatic hydrocarbons (PAH), aromatic hydrocarbons (BTEX), phenols, chlorinated hydrocarbons (CHC), the other inorganic compounds, and asbestos are named as the other significant contaminants for soil (URL 1, 2011).

In the light of the information presented above, a table that includes the potential sources of soil pollution and the priority soil pollutants arising from these sources was developed. For this purpose, various references (Ünlü et al., 2009; TMOEUA, 2010; Industry Profile Reports produced by UK Environment Agency (EA); URL 2, 2011; Demetriades, 2007) were studied to identify the potential soil pollutants associated with soil polluting activities in Turkey. The potentially soil polluting activities important for Turkey were listed with respect to their NACE Codes and potential soil pollutants (or groups of pollutants) for each activity were identified. Furthermore, the chemical substances listed in the regulations of other countries (i.e., Germany, France, Norway, Canada, Spain, Netherlands, and US EPA) were reviewed, since the chemicals given in the regulations represent the most commonly observed soil pollutants of a country. For example, the US EPA's list represents the commonly found contaminants at sites on the National Priorities List (NPL), which designates the sites having priority among the known contaminated sites throughout the US and its territories (US EPA, 1996b).

The priority soil pollutants for Turkey are identified based on the information gained from the abovementioned detailed literature search, as well as the compounds given in “the priority hazardous contaminants list” of Corbitt (1990). The twelve compounds being in subject of Stockholm Agreement on “Persistent Organic Polluters (POPs)”, most of which (DDT, heptachlor, chlordane, aldrin, dieldrin, endrin, toxaphene, mirex, PCB, hexachlorobenzene, dioxins and furans) have been included in regulations of the other countries, were also considered for preparation of the priority pollutants list for Turkey. Since SQSs are calculated by using chemical-specific toxicological data, the presence of toxicity data for each chemical was also assessed. The chemicals, for which no toxicological data is available, removed from the priority pollutants list. On the other hand, in some of the countries the soil quality criteria are calculated based on the groups of compounds instead of individual compounds, such as BTEX (benzene, toluene, ethylbenzene and xylene), PAHs (polyaromatic hydrocarbons), PCBs (polychlorinated biphenyls), HCHs (hexachlorocyclohexanes), chlorobenzenes, organotin compounds, dioxins and furans. However, for some of these compounds, no toxicological data were found in literature (e.g., BTEX, PAHs, HCHs, etc.). SQSs for these compounds are estimated by different methods (e.g. toxicological equivalence factor (TEF), or toxicological similarity to another substance), a detailed discussion on the calculation of SQSs for these special case chemicals are presented in Section 2.3.2. As a result, the priority soil pollutants for Turkey were identified as given in Table 2.

Table 2
Priority soil pollutants considered for the calculation of SQSs for Turkey.

Organics		
Acenaphthene	1,4-Dichlorobenzene	Naphthalene
Acetone (2-Propanone)	3,3'-Dichlorobenzidine	Nitrobenzene
Acrolein	1,1-Dichloroethane	2-Nitrophenol
Acrylamide	1,2-Dichloroethane	4-Nitrophenol
Acrylonitrile	1,1-Dichloroethylene	N-Nitrosodimethylamine
Aldrin	1,2-cis-Dichloroethylene	N-Nitroso-di-N-propylamine
Anthracene	1,2-trans-Dichloroethylene	N-Nitrosodiphenylamine
Atrazine	2,4-Dichlorophenol	PCBs (low risk and persistence)
Benz(a)anthracene	2,4-Dichlorophenoxy acetic acid	PCBs (high risk and persistence)
Benzene	1,2-Dichloropropane	Pentachlorobenzene
Benzidine	1,3-Dichloropropene	Pentachlorophenol
Benzo(a)pyrene	Dieldrin	Phenol
Benzo(b)fluoranthene	Diethylphthalate	Pyrene
Benzo(k)fluoranthene	2,4-Dimethylphenol	Pyridine
Benzoic acid	Dimethylphthalate	Styrene
Bis(2-chloroethoxy)methane	Di-n-butyl phthalate	1,2,4,5-Tetrachlorobenzene
Bis(2-chloroethyl)ether	4,6-Dinitro-o-cresol	2,3,7,8-TCDD
Bis(2-ethylhexyl)phthalate	2,4-Dinitrophenol	1,1,2,2-Tetrachloroethane
Bis(chloromethyl)ether	2,4-Dinitrotoluene	Tetrachloroethylene
Bromodichloromethane	2,6-Dinitrotoluene	Tetraethyllead
Bromoform	Di-n-octyl phthalate	Toluene
Butanol	1,2-Diphenylhydrazine	Total Petroleum Hydrocarbons (Aromatic) (EC9> - EC16) ^a
Butyl benzyl phthalate	Endosulfan	Total Petroleum Hydrocarbons (Aliphatic) (EC16> - EC35) ^a
Carbaryl	Endrin	Total Petroleum Hydrocarbons (Aliphatic) (EC5 - EC8) ^a
Carbazole	Ethylbenzene	Total Petroleum Hydrocarbons (Aliphatic) (EC8> - EC16) ^a
Carbofuran	Fluoranthene	Total Petroleum Hydrocarbons (Aromatic) (EC16> - EC35) ^a
Carbon disulfide	Fluorene	Total Petroleum Hydrocarbons (Aromatic) (EC5 - EC9) ^a
Carbon tetrachloride	Furan	Toxaphene
Chlordane	Heptachlor	Tributyltin oxide
p-Chloroaniline	Heptachlor Epoxide	1,2,4-Trichlorobenzene
Chlorobenzene	Hexachloro-1,3-butadiene	1,1,1-Trichloroethane
Chlorodibromomethane	Hexachlorobenzene	1,1,2-Trichloroethane
Chloroform	α -HCH (α -BHC)	Trichloroethylene
Chloromethane	β -HCH (β -BHC)	2,4,5-Trichlorophenol
beta-Chloronaphthalene	γ -HCH (Lindane)	2,4,6-Trichlorophenol
2-Chlorophenol	Hexachlorocyclopentadiene	Vinyl acetate
Chrysene	Hexachloroethane	Vinyl chloride (chloroethene)
m-Cresol	Hydroquinone	Xylene, mixture
o-Cresol	Indeno(1,2,3-cd)pyrene	m-Xylene
p-Cresol	Isophorone	o-Xylene
Cyclohexanone	Maneb	p-Xylene
DDD	MCPA	
DDE	Methoxychlor	
DDT	Methyl bromide	
Dibenz(a,h)anthracene	Methyl Tert-Butyl Ether (MTBE)	
1,2-Dichlorobenzene	Methylene chloride	
Inorganics		
Antimony	Cobalt	Thallium
Arsenic	Copper	Tin
Barium	Lead	Titanium
Beryllium	Mercury	Vanadium
Cadmium	Molybdenum	Zinc
Chromium (III)	Nickel	Cyanide
Chromium (total)	Selenium	Thiocyanate
Chromium (VI)	Silver	

^a EC: equivalent carbon number.

2.1.4. Determination of the key elements for derivation of Turkish SQSs

The approach for derivation of Turkish SQSs was determined based the regulatory needs for SQSs. For this purpose, the experience and scientific findings of the other countries were utilized. As explained before, in most of the European countries, Canada and the US, human health risk based soil criteria have been in use. On the other hand, some of these countries (e.g., Canada, the Netherlands, Spain, etc.) have already developed their soil quality criteria also to protect ecological receptors and some of the countries are currently working to develop ecological soil quality criteria. Being a relatively new field of interest compared to human health risk assessment, ecological risk assessment requires a broad knowledge on exposure assessment and dose-response assessment

performed on ecological receptors. Hence, development of ecological soil quality criteria is another area of concern requiring country specific data on potential ecological receptors, which is currently not available for Turkey. Therefore, the priority was given to the development of human health risk based SQSs.

Although the same principles are utilized for derivation of human health risk based SQSs, different countries end up with different soil quality criteria and. As mentioned before, one of the reasons for this situation is the differences in the intended use of (the role of) the SQSs, since they are used to pose different levels of risk (i.e., negligible, intermediate, and unacceptable risk). For this reason, the role of the SQSs within the contaminated site management system has to be defined explicitly

Table 3
Accepted risk levels by different organizations/countries (DEFRA, 2006).

Organization/ country	Context	Tolerable risk
WHO	Guidelines for Drinking Water Quality	10^{-5}
EU	Drinking Water Standards	10^{-6}
US EPA	Soil Screening Levels	10^{-6} (10^{-4} for additive risks)
RIVM	Dutch Intervention Values for Soil	10^{-4}
Norway	General	10^{-5}
Germany	General	10^{-5} (5×10^{-5} for additive risks)

from the beginning. In the framework of the developed contaminated sites management system for Turkey (Ünlü et al., 2009; TMOEUA, 2010), the role of the SQSs was defined as the generic standards used to eliminate sites that do not need further attention and to identify sites that require further site-specific risk assessment (SQSs correspond to intermediate risk level).

Although the general approaches of different countries were approximately known, essential details of only a few of these countries' calculation procedures for derivation of SQSs were reported in the literature. Nevertheless, the available documents, reports, regulatory standards and guidelines of the countries having well developed human health risk assessment procedures (i.e., the US EPA, Canada, Germany, the Netherlands, and Norway) were studied and compared with each other. In fact, the algorithms used for derivation of human health risk based SQSs are similar, the differences in the approaches result in derivation of different numerical values for SQSs. Therefore, it was decided to follow US EPA's general approach for development of Turkish SQSs due to its comprehensive documentation and availability of reports in English, as well as its development based on long-term practice and experience gained in this area. Besides, it should be noted that, the approaches of the other countries were also utilized for derivation of Turkish SQSs. For example; US EPA's, Canadian and Norwegian approaches were compared for determination of a generic dilution factor to calculate generic SQSs for migration to groundwater pathway. The Norwegian approach was adopted for comparison of SQSs developed with the background soil concentrations and detection limit of a chemical of concern and adjusting the SQSs with respect to those values. For derivation of the air dispersion factor (used to calculate SQSs for inhalation of volatiles and dusts) for Turkey, the approaches of both the US EPA and the UK Environmental Agency (EA) were examined and appropriate values were specified by comparison.

With regard to US EPA's approach, the generic SQSs for Turkey were derived based on the US EPA's *reasonable maximum exposure (RME) assumption* (US EPA, 1989), where human receptors in a residential setting intake reasonable amounts of contaminant for a reasonably conservative duration (US EPA, 1991). The site-specific parameters used in SQSs derivation were specified with respect to the average or typical soil characteristics and meteorological conditions of Turkey. In the following sub-sections, the key elements for derivation of SQSs, which include the studies performed for determination of a target (acceptable) risk level, the generic exposure scenarios and pathways considered, the generic values used for exposure parameters and contact rates, derivation of generic soil characteristics for Turkey and compilation of physical-chemical and toxicological data for derivation of Turkish SQSs, are explained.

Determination of the target (acceptable) risk level: Soil quality criteria for non-threshold (carcinogenic) substances are calculated based on a target (acceptable) risk level. In fact, target risk level does not only designate the perception of risk, but also a political decision made by authoritative bodies. Since SQSs are derived depending on the target risk level, the number of sites that are considered to be polluted is correlated with the target risk level defined. Consequently, it affects the financial resources allocated for assessment and/or remediation of a site.

As mentioned before, the target risk level for non-threshold compounds varies from 10^{-4} to 10^{-6} at different countries (see Table 3).

Considering the risk levels accepted by these organizations/countries, together with the opinion of the Turkish Ministry of the Environment, a conservative target risk level was set for Turkey to derive the generic SQSs. As a result, the target risk level was determined as 10^{-6} , meaning one-in-a-million excess lifetime cancer incidence. However, 10^{-6} risk level might be considered to be more strict compared to most of the European countries, it is completely related to the purpose of use (the role) of the soil criteria within the contaminated sites management system. Turkish SQSs developed are intended to be used for screening the sites that do not need further attention, or for highlighting the sites that need further site investigation. It is believed that this risk level would be helpful for screening the sites having pollutant concentrations below generic SQSs from further investigation without any doubt. On the other hand, it should be kept in mind that combination of any two or more pathways increases the health effect of a carcinogenic substance, and thus increases target risk level. For instance, 10^{-6} target risk level accepted for two (multiple) pathways may results in a risk level in the order of 10^{-5} , as the carcinogenic health effects are considered together. In most of the countries, combined effects of pollutants are considered, which may be reason for lower risk levels accepted in those countries.

Therefore, a target risk level of 10^{-6} is considered to be appropriate for Turkey while developing pathway-specific SQSs. Since the developed risk-based contaminated site management system has recently been enforced, the level of target risk level might be re-evaluated in the future, with respect to the requirements of the regulation, or based the experience to be gained in this field.

Determination of the generic exposure scenarios: For derivation of generic SQSs for Turkey, the residential land use scenario was considered to be a conservative scenario, for which human receptors are more susceptible to exposure to soil pollutants. Residential land use assumption was also reasonable for protection of the sensitive receptors (i.e., children and adults). However, the generic SQSs developed based on the residential land use assumption would be too conservative for some industrial sites where residential settings are not in the area of influence. Owing to this reason, the US EPA approach was taken as the basis and commercial/industrial land use scenario was also considered for derivation of Turkish SQSs. For industrial/commercial land use, two types of receptors are of concern; outdoor workers and indoor workers (US EPA, 2002). Since different exposure pathways and exposure parameters apply for these receptors, the generic SQSs derived for commercial/industrial land use-outdoor worker and commercial/industrial land use-indoor worker are different.

Some of the countries established their soil criteria based on land use types (e.g., Canada, Germany, Austria, Belgium, etc.), whereas the US EPA present pathway-specific soil criteria for the concerned exposure scenarios (i.e., residential, commercial/industrial). For development of Turkish generic SQSs, pathway specific soil criteria were regarded as more appropriate approach due to its comparative use during sampling practices. In order to measure contaminant levels in soil, two kinds of soil sampling strategy are performed; surface soil sampling (addresses ingestion, dermal and inhalation of fugitive dust pathways) and subsurface soil sampling (addresses inhalation of volatiles and migration to groundwater pathways) (US EPA, 1996a). In this regard, pathway specific soil criteria are advantageous allowing the comparison of surface and subsurface soil concentrations with the allowable soil concentra-

Table 4
Exposure parameters required for derivation of SQSs (US EPA, 1989; UK EA, 2009).

Parameters	Significance
EF (exposure frequency)	Represents the number of days per year in which a daily exposure event is considered to occur.
ED (exposure duration)	Refers to the length of time in years that a critical receptor assumed to be exposed to contaminant. Exposure frequency and exposure duration are used to estimate the total time of exposure.
EV (event frequency)	Refers to the number of events expected to occur per day.
IR (ingestion rate of soil)	Provides information on the amount of soil ingested on a daily basis.
IR_w (ingestion rate of groundwater)	Provides information on the amount of groundwater ingested on a daily basis.
SA (skin surface area exposed)	Refers to the surface area of the skin that is open for dermal contact.
AF (skin-soil adherence factor)	Provides information on the amount of soil adhered to, or in intimate contact with the skin, over the contact period for a single event.
BW (body weight)	Refers to average body weight over the exposure period. For exposures occurring during childhood years, average child body weight is used. For exposures occurring throughout the lifetime, age adjusted exposures are calculated.

tions for each pathway of concern. Furthermore, pathway specific SQSs allow for particular cases to disregard the soil quality criteria given for that pathway and to exclude the pathway that is not of concern from further investigation. In this respect, the main exposure pathways of the US EPA listed below were taken into consideration for derivation of Turkish human health risk based SQSs: (i) combined ingestion-dermal contact pathway, (ii) outdoor inhalation of fugitive particulates, (iii) outdoor inhalation of volatiles, and (iv) ingestion of groundwater (migration to groundwater).

The abovementioned exposure pathways are also considered by most of the countries, such as Germany, Norway, Canada, the UK, and others. However, for Belgium (Flanders and Walloon), Finland and the Netherlands ingestion of contaminated groundwater is not a generic pathway of concern, due to infrequent use of groundwater as potable water. Instead, *drinking water contaminated by permeation through pipes* is considered. However, ingestion of contaminated groundwater is an important pathway for Turkey, due to the frequent use of groundwater supplies in the country.

As a consequence, three different land use scenarios were considered for development of generic human health risk based SQSs for Turkey. Exposure pathways and receptors of concern for each of these land use scenarios are summarized below:

- **Residential Scenario:** In this scenario, children and adults are considered as the potential receptors. Ingestion-dermal contact, inhalation of volatiles, inhalation of fugitive particulates and migration to groundwater are considered as the main exposure pathways.
- **Commercial/Industrial Scenario - Outdoor Worker:** In this scenario, adults are considered as the potential receptors. Ingestion-dermal contact, inhalation of volatiles, inhalation of fugitive particulates and migration to groundwater are considered as the main exposure pathways.
- **Commercial/Industrial Scenario - Indoor Worker:** In this scenario, adults are considered as the potential receptors. Ingestion and migration to groundwater are considered as the main exposure pathways.

No other land use scenarios were considered within generic risk assessment because derivation of generic soil criteria for other land use scenarios, such as agricultural land use scenario and construction exposure scenario, involves many other variables to define exposure conditions. For instance, in agricultural land use scenario, human uptake from raised and consumed farm products and human exposure to contaminants through consumption of beef, milk and vegetables have to be estimated (Kerr et al., 1998) by use of chemical-specific plant uptake rates. However, such kind of information and data to be used for calculation of human uptake rate is lacking for Turkey. Similarly, for

some of the exposure pathways, it is hard to identify default parameters for Turkey. For example, *inhalation of volatiles in indoor air pathway* requires inputs such as building sizes, floor crack area, air exchange rate, etc. (US EPA, 2002; CCME, 2006), for which currently no comprehensive data available to define the typical conditions for Turkey. Likewise, for pathways, such as *consumption of homegrown products* or *consumption of meat, dairy or fish*, it is also very difficult to make a generalization for the whole country and to specify standardized values for the parameters, because at every region of Turkey, people has different living standards and consumption rates differ depending on the environment they live.

On the other hand, with respect to the contaminated sites management system of the new Regulation on Soil Pollution Control and Sites Contaminated with Point Sources (TMOEUA, 2010), generic SQSs are aimed to be used during generic risk assessment phase. In case additional human exposure pathways are identified, then site-specific risk assessment would be performed for that site. Thus, specified exposure scenarios and pathways are considered to be protective for the sites that do not include additional pathways.

Determination of the exposure parameters and contact rates: Human exposure to pollutants is expressed in terms of the intake of mass of substance per unit body weight per unit time (mg/kg-day) (US EPA, 1989). Thus, it is necessary to define the maximum exposure dose causing no adverse health effect and the human exposure conditions in order to derive human health risk-based SQSs. Exposure parameters that are used to describe the exposure conditions are presented in Table 4 together with explanations of their purpose of uses.

Even though the exposure parameters do not differ much from country to country, they are defined based on conservative values (e.g., US EPA) or based on national statistical data (e.g., Canada, UK) that describe the typical exposure conditions. US EPA recommends the use of reasonable conservative estimates for exposure parameters, in case necessary statistical data are not available (US EPA, 1989). With this concern, US EPA has selected the exposure parameters with respect to the RME assumption, which is based on “the highest exposure that is reasonably expected to occur at a site” (US EPA, 1989, 1996a). Thus, the default values for these variables represent a conservative situation regarding the characteristics of the potentially exposed population (US EPA, 1989). US EPA’s RME assumption was considered to be applicable also for derivation of Turkish SQSs; thus, relevant default values for the exposure parameters were used to calculate Turkish SQSs. The generic values for the exposure parameters and contact rates used for calculating Turkish SQSs are given in Table 5 with respect to the exposure scenarios considered.

Identification of the chemical-specific data requirement: Calculation of Soil Quality Standards require the knowledge of contaminant specific

Table 5

The generic values for the exposure parameters and contact rates used for derivation of Turkish SQSs (US EPA, 2002).

Parameters	Residential land use	Commercial/industrial land use		Unit
		Outdoor worker	Indoor worker	
EF (exposure frequency)	350	225	250	days/year
ED (exposure duration)	30 (6 for non-carcinogenic effects) ^a	25	25	years
EV (event frequency)	1	1	NA	events/day
IR_{soil} (ingestion rate of soil – child)	200	NA	NA	mg/day
IR_{soil} (ingestion rate of soil – adult)	100	100	50	mg/day
IR_w (ingestion rate of groundwater – child)	1	NA	NA	L/day
IR_w (ingestion rate of groundwater – adult)	2	2	2	L/day
InhR (inhalation rate) ^b	20	20	20	m ³ /day
SA (skin surface area exposed – child)	2800	NA	NA	cm ²
SA (skin surface area exposed – adult)	5700	3300	NA	cm ²
AF (skin-soil adherence factor – child)	0.2	NA	NA	mg/cm ² -event
AF (skin-soil adherence factor – adult)	0.07	0.2	NA	mg/cm ² -event
BW (body weight – child)	15	NA	NA	kg
BW (body weight – adult)	70	70	70	kg
LT (lifetime)	70	70	70	years

^a A child is defined as an individual between one and six years of age.^b Residential inhalation exposure to children and adults are evaluated by using the *RfC* toxicity criterion, which is based on an inhalation rate of 20 m³/day. No comparable toxicity criterion specific to childhood exposures is currently available.**Table 6**

Chemical-specific properties required to derive generic SQSs and the significance of these properties (US EPA, 1989).

Parameters	Unit	Significance
K_{oc} (soil organic carbon partition coefficient)	L/kg	Refers to chemical partitioning between organic carbon and water at equilibrium. The higher the K_{oc} , the more likely a chemical is to bind to soil or sediment than to remain in water.
K_d (soil-water partition coefficient)	L/kg	Refers to chemical partitioning between soil and water. The higher the K_d , the more likely a chemical is to bind to soil or sediment than to remain in water.
H' (Henry's law constant)	unitless	Refers to chemical partitioning between air and water at equilibrium. The higher the Henry's Law constant, the more likely a chemical is to volatilize than to remain in water.
S (solubility in water)	mg/L	Refers to an upper limit on a chemical's dissolved concentration in water at a specified temperature.
D_i (diffusivity in air)	cm ² /s	Refers to the movement of a molecule in a gas medium as a result of differences in concentration. The higher the diffusivity, the more likely a chemical is to move in response to concentration gradients.
D_w (diffusivity in water)	cm ² /s	Refers to the movement of a molecule in a liquid as a result of differences in concentration. The higher the diffusivity, the more likely a chemical is to move in response to concentration gradients.
MP (melting point)	°C	Refers to the temperature at which the physical state of chemical changes from solid to liquid. Used to determine the physical state of organic chemicals at typical soil temperatures
ABS_{GI} (gastro intestinal absorption factor)	unitless	Used to adjust the oral reference dose (RfD) and cancer slope factor (SF) for a contaminant to dermal dose. If gastrointestinal absorption is greater than 50%, no adjustment is made.
ABS_d (dermal absorption factor)	unitless	Refers to the average dermal absorption values across a range of soil types, loading rates, and chemical concentrations.

physical-chemical and toxicological properties. Relevant data were compiled from available databases as described in the following subsections.

Compilation of physical-chemical data

Once contaminants are released into the environment, their physical-chemical characteristics play a significant role in determining their environmental fate and mobility. Therefore, various physical-chemical properties of contaminants should be known in order to estimate the concentrations of chemicals in different environmental media (i.e., air, water and soil).

With respect to the exposure pathways considered in the generic scenario, the chemical-specific properties required to derive generic SQSs and the significance of these properties are given in Table 6. Among these parameters, K_{oc} (soil organic carbon partition coefficient), K_d (soil-water partition coefficient), H' (Henry's law constant), S (solubility in water), D_i (diffusivity in air), and D_w (diffusivity in water) are used for estimating the volatilization factor (VF), saturation concentration (C_{sat}) and the partitioning between soil and groundwater. Melting point (MP) of a contaminant should be known in order to define the physical state of contaminant at typical soil temperatures. Physical state of the contaminant is important for assessing the existence of free liquid phase (i.e., non-aqueous phase liquid, NAPL) contamination in soil. Hence, these

parameters are required to derive the SQSs for inhalation of volatiles and migration to groundwater pathways.

The available sources of information were closely examined for compilation of the physical-chemical parameters required for the derivation of SQSs. The Risk Assessment Information System (RAIS) online database (URL 3, 2011) was determined as a data-rich platform to gather the chemical-specific data needed. The RAIS online database, which includes toxicological and chemical-specific data, is developed by Oak Ridge National Laboratory (ORNL) in 1996 with the support of the US Department of Energy (URL 3, 2011). The RAIS provides the risk assessment tools, such as guidance documents, tutorials, databases, historical information and risk models, to users from 45 State Governments, many Federal agencies and over 60 countries (URL 3, 2011). Besides, many universities and research institutes use the RAIS as the prime source of information for their risk assessment activities. In the RAIS, the main source of chemical-specific information is EPI Suite (the database program developed by the US EPA) (URL 2, 2011) and all information presented by the RAIS complies with the US EPA guidance (URL 3, 2011). Therefore, it was considered as a reliable source for providing the chemical-specific data to be used in derivation of Turkish SQSs. However, RAIS presents K_{oc} (for organics) and K_d (for inorganics)

Table 7
The chemical-specific toxicological data used for derivation of generic SQSs.

Parameters	Unit	Significance
SF_o (oral slope factor)	$(\text{mg/kg-d})^{-1}$	Refers to an upper-bound on the increased cancer risk from a lifetime exposure to an agent by ingestion. This estimate, usually expressed in units of proportion (of a population) affected per mg of substance/kg body weight-day.
RfD_o (oral reference dose)	mg/kg-d	Refers to an estimate of a daily oral exposure to human population that is likely to be without an appreciable risk of deleterious effects during lifetime and expressed as expressed in units of mg of substance/kg body weight-day.
URF (inhalation unit risk factor)	$(\mu\text{g}/\text{m}^3)^{-1}$	Refers to the upper-bound excess lifetime cancer risk estimated to result from continuous exposure to an agent at a concentration of $1 \mu\text{g}/\text{m}^3$ in air.
RfC (inhalation reference concentration)	mg/m^3	Refers to an estimate of a continuous inhalation exposure to human population that is likely to be without an appreciable risk of deleterious effects during lifetime.
C_w (target soil leachate concentration)	mg/L	Refers to the allowable maximum concentration level for drinking water.

only at a pH of 6.8 that is compatible with US EPA's methodology. For this reason, K_{oc} and K_d values for the pH dependent contaminants were compiled from EPI Suite (URL 4, 2011; US EPA, 2002).

As a result, the physical-chemical property values for more than 800 substances were compiled in the MS Excel based data library to facilitate the access to the information required for development of SQSs. This library was integrated with the computational tool (details of which will be presented in a separate paper) that was developed for calculating generic and site-specific SQSs. Thus, any upgrade made in chemical-specific values can be monitored from the RAIS and reflected to the data library of the computational tool to update SQSs accordingly.

Compilation of toxicological data

Risk based SQSs are derived based on toxicological data. The chemical-specific toxicological data, which defines the human health benchmarks, used for derivation of generic SQSs are listed in Table 7. Oral reference dose (RfD_o) and inhalation reference concentration (RfC) are used to estimate chronic non-carcinogenic health effects, while oral slope factor (SF_o) and inhalation unit risk factor (URF) are used to estimate risks for carcinogenic effects (URL 6, 2011). SQSs for ingestion and inhalation pathways are derived using of these toxicological parameters.

Since the toxicological data presented in the RAIS database are compatible with the US EPA methodology for development of soil quality criteria (URL 3, 2011), the toxicity data used for derivation of Turkish SQSs were obtained from the RAIS. This database contains toxicity information for more than 1000 chemicals, which are reviewed and updated regularly.

The toxicological data presented by the RAIS are gathered from the US EPA's Integrated Risk Information System (IRIS), EPA's Provisional Peer Reviewed Toxicity Values (PPRTVs), and other sources, such as the California EPA (CalEPA), the Agency for Toxic Substances and Disease Registry (ATSDR) and the Health Effects Assessment Summary Tables (HEAST) in accordance with the hierarchy stated in the OSWER Directive 9285.7-53 (US EPA, 2003a). Any upgrade in the toxicological values is indicated in the RAIS database separately, thus it is easy to monitor the latest amendments and transfer them to the data library of the computational tool to update SQSs accordingly.

On the other hand, target leachate concentration in soil (C_w) is used for derivation of the SQSs for migration to groundwater pathway. Because groundwater is used frequently for drinking purposes in Turkey, drinking water standards given in TS-266 Water Intended for Human Consumption (TSE, 2005) published by Turkish Standardization Institute were accepted as the target soil leachate concentration. For the compounds that are not included in TS-266, drinking water standards of World Health Organization (WHO, 2008) were used. For the compounds that are not included in any of these standards, health based limits (HBL), which are calculated based on 10^{-6} target risk level or a Hazard Quotient (HQ) of 1, were used. Details of Soil Quality Standards calculation procedure are given in Section 2.3.

2.2. Derivation of generic site characteristics

As the environmental characteristics of a site control the fate and transport of contaminants in different environmental media, it is essential to determine the generic site characteristics for derivation of country-specific SQSs. The generic site characteristics important for derivation of SQSs include meteorological properties (e.g., temperature, precipitation, wind speed and direction), soil characteristics (e.g., soil pH, soil texture, soil organic carbon-SOC content), vegetative cover properties, and hydrogeologic properties of underlying aquifers (e.g., hydraulic conductivity, hydraulic gradient, aquifer thickness) (US EPA, 1989). For this reason, the generic site characteristics that are representative of the country-specific properties and conservative for a range of site conditions need to be specified for derivation of generic SQSs. The role of hydrogeologic properties of underlying aquifers are addressed in Section 2.3.1 "Migration to groundwater" pathway (see Eqs. (16.5) and (16.6) of Table 16b). In the following section, specification of the generic soil characteristics and other relevant generic site characteristics related to derivation of air dispersion factors are discussed.

2.2.1. Specification of the generic soil characteristics

In order to calculate pathway-specific SQSs by use of US EPA approach, generic subsurface soil characteristics, such as soil organic matter (SOM) content, soil pH, fraction of organic carbon in soil (f_{oc}), and parameters that depend on soil texture (e.g., dry soil bulk density (ρ_b), soil particle density (ρ_s), total soil porosity (n), water filled soil porosity (θ_w), air filled soil porosity (θ_a)) are needed (See Table 8). These parameters are used to address exposure to pollutants by inhalation of volatiles migrating from soil to ambient air, ingestion of contaminated groundwater and to calculate the soil saturation limit (C_{sat}), which is an indicator for potential existence of free phase contaminant in soil. The same parameters (i.e., f_{oc} , ρ_b , ρ_s , n , θ_w , and θ_a) are also needed to represent the generic subsurface soil characteristics used to estimate contaminant release into groundwater, thus, SQSs for ingestion of groundwater pathway. Moreover, for determination of the concentration of pollutants in soil solution, soil-water partition coefficient, K_d , (for organics $K_d = K_{oc} \times f_{oc}$ where K_{oc} being organic-carbon partition coefficient) is used. K_d depends on chemical properties, as well as the characteristics of soil (e.g. SOM content, soil pH).

For specification of the generic soil characteristics (i.e., soil texture, pH and f_{oc}) for Turkey, nationwide soil characteristics for soil pH, SOM content and texture that are determined based on the analyses of more than 243,000 soil samples (Eyüpoğlu, 1999) was utilized. According to the results of analyses, 62% of Turkish soils are slightly alkaline (pH between 7.5 and 8.5), which can be explained by the calcareous parent material of underlying soils in considerable part of the country (Eyüpoğlu, 1999). For this reason, the generic pH value of 6.8 defined by US EPA was considered to be not applicable for Turkey.

Table 8

Generic soil characteristics and their numerical values needed to be specified for derivation of Turkish SQSs.

Parameters	Significance	
pH (soil pH)	Affects mobility of metals and ionizing organics, because K_d and K_{oc} of some substances change as a function of pH.	
f_{oc} (soil organic carbon fraction)	Related to the soil organic matter content which determines the phase partitioning of contaminants.	
ρ_b (dry soil bulk density)	Measure of the dry-weight of the soil per unit volume.	
ρ_s (soil particle density)	Measure of the weight of the soil solids only per unit volume.	
n (total soil porosity)	Refers to the fraction of bulk soil volume occupied by pores (i.e., by air and water).	
θ_w (water filled soil porosity)	Refers to the amount of soil pore space occupied by water.	
θ_a (air filled soil porosity)	Refers to the amount of soil pore space occupied by air.	
	Value	Unit
pH (soil pH)	6.7 and 8.2	-
f_{oc} (soil organic carbon fraction)	0.006 ^a	g/g
ρ_b (dry soil bulk density)	1.5	kg/L
ρ_s (soil particle density)	2.65	kg/L
n (total soil porosity)	$1 - (\rho_b / \rho_s)$	cm ³ /cm ³
θ_w (water filled soil porosity)	0.15 ^b	cm ³ /cm ³
θ_a (air filled soil porosity)	$n - \theta_w$	cm ³ /cm ³

^a To be conservative, fraction of soil organic carbon is taken as 0.006 g/g for inhalation pathway, whereas it is taken as 0.002 g/g for migration to groundwater pathway.

^b To be conservative, water filled soil porosity is taken as 0.15 for inhalation pathway, whereas it is taken as 0.30 for migration to groundwater pathway.

In order to set a conservative pH value, the effect of pH on fate and transport of chemicals has to be understood. Fundamentally, pH affects the chemical partitioning between soil and water phases, which is defined by K_d (for organics $K_d = K_{oc} \times f_{oc}$). However, pH affects the mobility of metals and ionizing organics in different ways. K_{oc} shows the chemical partitioning between organic carbon and water at equilibrium and is used in calculation of SQSs for inhalation of volatiles and migration to groundwater pathways. Unlike non-ionizing hydrophobic organic compounds, soil-water partitioning behavior of ionizing organics are affected by soil pH. The ionizing organic compounds such as amines, carboxylic acids, and phenols exhibit different sorption behaviors under different pH conditions (US EPA, 1996a). For ionizing organic compounds, K_{oc} decreases (the chemicals tend to remain in water, instead of binding to soil) with the increasing soil pH. Since volatilization and in turn SQSs are directly proportional to K_{oc} values, higher pH values (lower K_{oc} values) result in generation of more conservative SQSs for inhalation of volatiles and migration to groundwater pathways. On the other hand, for metals, soil-water partition coefficient, K_d , is used to calculate SQSs for inhalation of volatiles and migration to groundwater. Although, K_d for metals is most sensitive to various geochemical parameters and processes, it is most affected by the changes in pH (US EPA, 1996a). However, all metals exhibit different behaviors with pH change. For example, K_d values for As, Ba, Be, Cd, Cr³⁺, Hg, Ni, Ag, Th, and Zn are directly proportional to pH, whereas K_d values for Cr⁶⁺ and Se are inversely proportional and for Sb and V, K_d does not depend on pH. Similar to ionizing organics, lower K_d values result in production of more conservative SQSs for inhalation of volatiles and migration to groundwater pathways.

Since the aim of Turkish generic SQSs is to screen sites that do not need further attention, a conservative pH value had to be selected. However, due to different behaviors of ionizing organics and metals in different pH conditions, it was not possible to define a generic pH value that produces conservative SQSs for both kinds of contaminants. Owing to this reason, it was decided to select different default pH values for ionizing organics and metals with respect to their response to pH change. For this purpose, the cumulative probability distribution of pH for Turkish soils was considered. The pH values corresponding to 10th and 90th percentiles were determined as 6.7 and 8.2, respectively. For ionizing organic compounds, Cr⁶⁺ and Se, a generic pH of 8.2; and for the other metals (i.e. As, Ba, Be, Cd, Cr³⁺, Hg, Ni, Ag, Tl, and Zn) a generic pH of 6.7 was selected and used for calculation of SQSs for inhalation of volatiles and migration to groundwater pathways.

As known, SOM content is an indication of the organic fraction in solid phase of soil. Natural factors, such as climate, soil parent material,

land cover and/or vegetation, topography; and human-induced factors, such as land use, management and degradation affect the SOM content (Jones et al., 2004). With regard to the SOM content of Turkish soils, about 65% of the soils contain very low or low SOM, which clearly indicates that soils of Turkey are generally poor in terms of organic matter content. Due to the extensive vegetative cover stemming from abundant rainfall, the Black Sea Region has a relatively high organic matter content compared to other regions. On the other hand, SOM content is lower in Central and Southeast Anatolia because of weak vegetative cover, high temperatures and limited amount of precipitation (Eyüpoğlu, 1999).

In order to determine a conservative generic f_{oc} value for Turkey, the relationship between SOM content and SQSs should be considered. As SOM content of soil increases, more pollutant will be adsorbed to solid fraction of soil, which means the pollutant will be less available in soil solution for plant uptake, vaporization or migration to groundwater (UK EA, 2009). Therefore, lower SOM content assumption results in more conservative scenarios regarding calculation of SQSs.

According to the report of Eyüpoğlu (1999), about 44% of the soils in Turkey contain low SOM (1–2%). In order to produce conservative SQSs, the generic value of SOM for Turkey is determined with respect to the lower end of this interval as 1%. Since soil organic carbon (SOC) is the major component of SOM (Jones et al., 2004), the fraction of SOM is related to the SOC as given below (US EPA, 1996a):

$$f_{om} = 1.724 f_{oc} \quad (1)$$

where f_{om} is the fraction of SOM content; and f_{oc} is the fraction of SOC content. With use of this equation, the fraction of organic carbon for Turkish soils was calculated as 0.6%. Actually, this value is compatible with the generic value defined by US EPA (1996a) for derivation of SQSs for inhalation of volatiles pathway. On the other hand, SOC content decreases with depth and the probable range for SOC content for subsurface soils is determined as 0.1% to 0.3% by US EPA (1996a). In this regard, the generic f_{oc} value for Turkey for subsurface soils was accepted as 0.2%, as it was also selected by US EPA (1996a).

The other soil parameters (i.e., ρ_b , ρ_s , n , θ_w , and θ_a), which depend on the soil texture, are all related to each other. Since volatilization is the most sensitive to water filled soil porosity, θ_w , which affects the air filled porosity and consequently the steady-state flux of volatile contaminants from soil (US EPA, 1996a), a conservative value (0.15) were defined for θ_w by US EPA (1996a). In fact, this value takes place between wilting point (0.09) and mean field capacity (0.20) given for loam soils having moderate hydrologic characteristics (US EPA, 1996a; Carsel and Parrish, 1988). For loam soil type, the mean porosity is determined as

Table 9

Generic site characteristics and their numerical values needed to be specified for derivation of Turkish SQSs.

Parameters	Significance
Q/C_{vol} (air dispersion factor for volatiles)	Used to estimate volatilization factor (VF) that relates the concentration of contaminant in soil with the concentration of volatiles in the air. Depends on the source size.
Q/C_{wind} (air dispersion factor for fugitive dusts)	Used to estimate particulate emission factor (PEF) that relates the concentration of contaminant in soil with the concentration of dust particles in the air. Depends on the source size.
V (fraction of vegetative cover)	Fraction of continuous vegetative cover. Used to estimate PEF .
U_m (mean annual wind speed at 10 m)	Annual average wind speed at 10m above ground. Used to estimate PEF .
U_t (equivalent threshold value of wind speed at 10 m)	Equivalent threshold value of wind speed at 10m. Used to estimate PEF .
Q/C_{vol} (air dispersion factor for volatiles)	Value Unit
Q/C_{wind} (air dispersion factor for fugitive dusts)	27.61 ^a g/m ² -s per kg/m ³
V (fraction of vegetative cover)	59.24 ^b g/m ² -s per kg/m ³
U_m (mean annual wind speed at 10 m)	0.2 unitless
U_t (equivalent threshold value of wind speed at 10 m)	3.0 m/s
$F(x)$ (function dependent on U_m/U_t derived using Cowherd et al. (1985))	8.28 m/s
	6.67×10^{-2} unitless

^a For residential scenario (source size: 0.01 ha) 27.61 g/m²-s per kg/m³; for industrial/commercial scenario (source size: 1 ha) 8.96 g/m²-s per kg/m³.

^b For residential scenario (source size: 0.01 ha) 59.24 g/m²-s per kg/m³; for industrial/commercial scenario (source size: 1 ha) 19.81 g/m²-s per kg/m³.

0.43 (US EPA, 1996a; Carsel and Parrish, 1988). In fact, water content of subsurface soil is always more than surface soil due to reduced evaporative losses from deeper depths. Therefore, a typical value of 0.30 for water content was used by US EPA to represent subsurface conditions. For loam soils, this value takes place between the mean field capacity (0.20) and saturated volumetric water content for loam (US EPA, 1996a; Carsel and Parrish, 1988). Considering the range of soil bulk density for surface soils (generally between 1.3 and 1.7 g/cm³), an average value of 1.5 g/cm³ was accepted by US EPA, which is also consistent with the soil porosity defined. Since the soil particle density for most soil mineral material is 2.65 g/cm³, it was accepted as generic value. More than 50% of the soils in Turkey have loam texture (Eyüpoğlu, 1999). Thus, the generic values defined by US EPA for ρ_b , ρ_s , n , θ_w , and θ_a are also applicable for Turkey, and these values are accepted as the generic values for Turkey (Table 8).

2.2.2. Specification of other generic site characteristics related to derivation of air dispersion factors

According to US EPA approach, exposure to contaminants by inhalation of fugitive dusts pathway is estimated by use of particulate emission factor (PEF) that denotes the relationship between soil and air phase contaminant concentrations resulting from particle suspension (US EPA, 2002). In fact, PEF refers to the annual average particulate matter emission resulting from wind erosion and given as the inverse of the amount of emissions per volume of air inhaled (m³/kg). Exposure to contaminants by inhalation of volatiles pathways is estimated by use of soil-to-air volatilization factor (VF) that relates the concentration of a contaminant in soil to the concentration of the contaminant in soil air resulting from volatilization, and it is expressed in m³/kg (US EPA, 2002).

Site characteristics such as fraction of continuous vegetative cover (V), mean annual wind speed (U_m), equivalent threshold value of wind speed (U_t) and air dispersion factor for fugitive dusts (Q/C_{wind}), which effects the dispersion of fugitive dust emissions in air, are used to calculate PEF . Similarly, VF refers to the annual average emission resulting from volatilization. VF is based on soil characteristics (i.e., f_{OC} , ρ_b , ρ_s , n , θ_w , and θ_a), chemical-specific parameters (K_d , D_i , D_w , and H') and the air dispersion factor for volatiles (Q/C_{vol}), which represents the dispersion of volatile emissions in air.

To estimate the emissions of volatiles and dusts and to derive SQSs for inhalation pathway, Q/C_{wind} and Q/C_{vol} are needed. Since these factors depend on the meteorological conditions and site characteristics, factors representing the regional conditions of the country should be used in the derivation of SQSs. The dispersion of fugitive dusts and

volatiles under the prevailing meteorological conditions can be estimated by use of an air dispersion models simulating the dispersion of pollutants in the atmosphere. Further details of the study performed for derivation of the generic values of the air dispersion factors (Q/C_{vol} and Q/C_{wind}), and specification of U_m , U_t , and V certainly justify to be discussed in a separate paper and are beyond the scope of this paper. The derived generic values of these parameters are presented in Table 9.

The generic air dispersion factors (Q/C_{wind} and Q/C_{vol}) for Turkey were derived and the generic values for the corresponding site characteristics (U_m , U_t , and V) used in calculation of PEF were specified. For this purpose; the approaches of the countries for derivation of soil quality criteria for inhalation of fugitive dusts and volatiles were reviewed. In some of the countries (e.g. Austria, Canada, Czech Republic, France, Norway, Sweden), inhalation of volatiles at outdoor is not considered among generic pathways and in some of the countries (e.g. Germany, Italy, the Netherlands) a generic value representing the amount of dust emissions per volume of air inhaled (mg/m³) is given, but the methodology used for derivation of these values is not presented in available documents/reports. The details of the methodology used by US EPA for derivation of air dispersion factors is presented in several documents (US EPA, 1996a; US EPA, 2002), and it is also utilized by the UK Environment Agency (UK EA, 2009) in conjunction with some revisions. To derive the generic values for the air dispersion factors (Q/C_{vol} and Q/C_{wind}) for Turkey, the differences between the approaches of US EPA (1996a, 2002) and UK EA (2002, 2009) were considered and advantageous parts of these two approaches were adopted (İpek, 2011). The US EPA air dispersion model, AERMOD (US EPA, 2009), was run by integration of 7 years of hourly meteorological data of seven stations distributed over Turkey (İpek, 2011). Air dispersion factors for each station were calculated and the minimum Q/C value (yielding maximum air concentration), which has been obtained for Adana station, was selected as the generic for Q/C_{vol} value to be used for derivation of SQSs for inhalation of volatiles pathway. In order to determine a generic value for Q/C_{wind} , the effects of other factors (i.e. U_m , U_t , V) influencing the actual air concentration of particulate emissions was considered. These parameters were identified with respect to the meteorological and vegetation cover characteristics of Turkey. All generic values of site characteristics used in calculations of SQSs are given in Table 9.

Because of the low mean annual rainfall, especially in the central part of Turkey, and because the lands allocated for industrial facilities are mostly not suitable for vegetation, a generic value of 0.2 was accepted for fraction of continuous vegetative cover. The mean annual wind speed (U_m) for each station was determined from 30 years of meteorological

Table 10
Generic land use scenarios and relevant pathways of concern.

Potential Receptor	Residential scenario child, adult	Commercial/industrial scenario: outdoor worker adult	Commercial/industrial scenario: indoor worker adult
Soil Ingestion	✓	✓	✓
Dermal Contact	✓	✓	-
Inhalation of Volatiles	✓	✓	-
Inhalation of Fugitive Particulates	✓	✓	-
Migration to Groundwater	✓	✓	✓

Table 11
Determination of the SQSs considering the carcinogenic risks and non-carcinogenic health effects of a contaminant.

Ingestion-dermal(mg/kg)		Inhalation of volatiles(mg/kg)		Inhalation of fugitive particulates (mg/kg)		Migration to groundwater(mg/kg)	
SQS ₁ ^c	SQS ₁ ^{nc}	SQS ₂ ^c	SQS ₂ ^{nc}	SQS ₃ ^c	SQS ₃ ^{nc}	SQS ₄ ^c	SQS ₄ ^{nc}
SQS ₁		SQS ₂		SQS ₃		SQS ₄	

*SQS^c (SQS calculated considering carcinogenic risks), SQS^{nc} (SQS calculated considering non-carcinogenic health effects); subscripts indicating the pathway; 1- soil ingestion/dermal contact, 2-inhalation of fugitive particulates, 3-inhalation of volatiles, 4-ingestion of contaminated groundwater.

bulletin published by the Turkish State Meteorological Works. To determine the threshold friction velocity, US EPA's conservative approach, which was also adopted by UK EA, was followed.

2.3. Exposure model for calculation of Soil Quality Standards

US EPA's "Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites" (2002), "Soil Screening Guidelines: Technical Background Document" (1996a) and "Soil Screening Guidance: User's Guide" (1996b) were the primary documents utilized for development of Turkish generic SQSs. The final versions of the risk-based equations that take place in US EPA's "Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites" (2002) are used for calculations. All of these equations are based on chronic exposures and reasonable maximum exposure (RME) assumption (US EPA, 1996b).

Generic SQSs were calculated for three different land use scenarios (residential scenario, commercial/industrial scenario: outdoor worker, and commercial/ industrial scenario: indoor worker) by use of US EPA's standardized sets of equations. The exposure pathways and the potential receptors considered in each of these land use scenarios are summarized in Table 10.

For each pathway (combined soil ingestion and dermal contact, inhalation of fugitive particles, inhalation of volatiles, and migration to groundwater), soil concentrations are calculated for the carcinogenic risks and for the non-carcinogenic health effects of chemicals. In other words, pathway specific soil concentrations of carcinogenic and non-carcinogenic chemicals that correspond to the target cancer risk level (10^{-6}) and target hazard quotient (1), respectively, are calculated for each compound. The lowest of these two concentrations (i.e., soil concentration calculated considering the carcinogenic risks and soil concentration calculated considering the non-carcinogenic effects of a substance) is set as the generic SQS. For example; SQS₁^c denoting the *carcinogenic* SQS and SQS₁^{nc} denoting the *non-carcinogenic* SQS for ingestion-dermal contact pathway were calculated and the lower of these two values is considered as the SQS₁ for ingestion-dermal contact pathway (where subscript indicating the pathway) as shown in Table 11.

2.3.1. Pathway-specific generic SQSs

The equations of the exposure model used for calculating the pathway specific SQSs are presented in the following sections. The exposure parameters and the contact rates are previously given for residential land use scenario. In order to calculate the SQSs for commercial/industrial land use scenarios, references should be made to Tables 5 and 10.

Soil ingestion and dermal absorption: The equations used to calculate SQS for combined soil ingestion-dermal absorption exposure pathway

are given in Tables 12a and 12b. Eq. (12.1) is used for exposure to carcinogenic pollutants. For carcinogenic compounds, duration of exposure is critical because the toxicity criteria are based on "lifetime average daily dose" (US EPA, 1996b). Hence, the total dose received is averaged over a lifetime of 70 years. Besides, the maximum exposure duration is assumed to be 30 years which is considered as the high-end period for an individual to live in the same residence from childhood to adulthood (US EPA, 1996b). Because exposure to soil is higher during childhood and decreases with age, time-weighted average soil ingestion rate (Eq. (12.2)) is used (US EPA, 1996b).

Due to the variation in skin surface area, skin-soil adherence factor and body weight for children and adults, age-adjusted dermal factor (SF_S) is used (See Eq. (12.3)). Eq. (12.5) is used to calculate SQSs for exposure to non-carcinogenic pollutants. In this equation, it is assumed that individuals are exposed to chemicals commonly during childhood by inadvertent ingestion of soil (US EPA, 1996b). Since no toxicity data are presently available to evaluate dermal exposures to chemicals, oral toxicity values are extrapolated by use of Eqs. (12.4) and (12.6).

Inhalation of fugitive particulates and volatiles: Inhalation risk from fugitive particulates results from contaminant concentrations in the surface soil; e.g., the top 2 cm of soil profile (US EPA, 1996a). On the other hand, the entire column of contaminated soil can contribute to volatile emissions at a site. Thus, contaminant concentrations in subsurface soils are of primary concern for quantifying the risk from volatile emissions (US EPA, 1996a). Owing to these reasons, different sampling strategies are used for surface soil and subsurface soil during generic risk analysis. Consequently, SQSs for inhalation of fugitive particulates and inhalation of volatile contaminants pathways are calculated using different equations. In the following sections, the equations used to calculate SQSs for inhalation of fugitive particulates and inhalation of volatiles are presented.

Inhalation of the fugitive particulates pathway is of concern for certain metals but does not appear to be of concern for organic compounds (US EPA, 1996a). Furthermore, for organic compounds, the SQSs calculated for ingestion-dermal absorption pathway is much stringent than the SQSs calculated for inhalation of fugitive particulates. Since both ingestion-dermal absorption and inhalation of fugitive particulates pathways are important for surface soils, SQS for ingestion-dermal absorption pathway is necessarily protective for this media (US EPA, 1996b). Therefore, SQSs for the fugitive particulates pathway are only presented for inorganic compounds. On the other hand, SQSs for the inhalation of volatiles pathway are not provided for inorganic compounds, because these chemicals are not volatile, mercury being an exception since it is volatile (US EPA, 1996a).

Table 12a

Equations used to calculate SQSs for soil ingestion-dermal contact pathway.

$SQS_1^c \text{ (mg/kg)} = \frac{TR \times AT \times 365d/yr}{(EF \times 10^{-6} \text{ kg/mg})(SF_o \times IF_{soil/adj}) + (SF_{ABS} \times SFS \times ABS_d \times EV)}$			(12.1)
where;			
SQS_1^c (carcinogenic SQS for soil ingestion-dermal absorption)	-	mg/kg	
TR (target cancer risk)	10^{-6}	unitless	
AT (averaging time)	70	year	
EF (exposure frequency)	350	day/year	
SF_{ABS} (dermally adjusted slope factor)	chemical-specific ^a	$(\text{mg/kg-day})^{-1}$	
SFS (age-adjusted dermal factor)	360 ^b	mg-year/kg-event	
ABS_d (dermal absorption factor)	chemical-specific ^c	unitless	
EV (event frequency)	1	event/day	
SF_o (oral slope factor)	chemical-specific ^c	$(\text{mg/kg-day})^{-1}$	
$IF_{soil/adj}$ (age-adjusted soil ingestion factor)	114	mg-year/kg-day	
^a See Eq. (112.4); ^b See Eq. (112); ^c See Toxicological Database.			
$IF_{soil/adj} = \left[\frac{IR_{soil/1-6} \times ED_{1-6}}{BW_{1-6}} \right] + \left[\frac{IR_{soil/7-31} \times ED_{7-31}}{BW_{7-31}} \right]$			(12.2)
where;			
$IF_{soil/adj}$ (age-adjusted soil ingestion factor)	-	mg-year/kg-day	
$IR_{soil/1-6}$ (ingestion rate of soil age 1-6)	200	mg/day	
$IR_{soil/7-31}$ (ingestion rate of soil age 7-31)	100	mg/day	
ED_{1-6} (exposure duration during ages 1-6)	6	year	
ED_{7-31} (exposure duration during ages 7-31)	24	year	
BW_{1-6} (average body weight from ages 1-6)	15	kg	
BW_{7-31} (average body weight from ages 7-31)	70	kg	
$SFS = \left[\frac{SA_{1-6} \times AF_{1-6} \times ED_{1-6}}{BW_{1-6}} \right] + \left[\frac{SA_{7-31} \times AF_{7-31} \times ED_{7-31}}{BW_{7-31}} \right]$			(12.3)
where;			
SFS (age-adjusted dermal factor)	-	mg-year/kg-event	
SA_{1-6} (skin surface area exposed-child)	2800	cm ²	
SA_{7-31} (skin surface area exposed-adult)	5700	cm ²	
AF_{1-6} (skin-soil adherence factor-child)	0.2	mg/cm ² -event	
AF_{7-31} (skin-soil adherence factor-adult)	0.07	mg/cm ² -event	
ED_{1-6} (exposure duration-child)	6	year	
ED_{7-31} (exposure duration-adult)	24	year	
BW_{1-6} (body weight-child)	15	kg	
BW_{7-31} (body weight-adult)	70	kg	

Table 12b

Equations used to calculate SQSs for soil Ingestion-dermal contact pathway (cont'd).

$SF_{ABS} = \frac{SF_o}{ABS_{GI}}$			(12.4)
where;			
SF_{ABS} (dermally adjusted slope factor)	-	$(\text{mg/kg-day})^{-1}$	
SF_o (oral slope factor)	chemical-specific ^a	$(\text{mg/kg-day})^{-1}$	
ABS_{GI} (gastro-intestinal absorption factor)	chemical-specific ^a	unitless	
^a See Toxicological Database.			
$SQS_1^{nc} = \frac{THQ \times BW \times AT \times 365 \text{ d/yr}}{(EF \times ED \times 10^{-6} \text{ kg/mg}) \left(\left(\frac{1}{RfD_o} \times IR \right) + \left(\frac{1}{RfD_{ABS}} \times AF \times ABS_d \times EV \times SA \right) \right)}$			(12.5)
where;			
SQS_1^{nc} (non-carcinogenic SQS for soil ingestion-dermal absorption)	-	mg/kg	
THQ (target hazard quotient)	1	unitless	
BW (body weight)	15	kg	
AT (averaging time)	6 ^a	year	
EF (exposure frequency)	350	day/year	
ED (exposure duration)	6	year	
RfD_o (oral reference dose)	chemical-specific ^b	mg/kg-day	
IR (soil ingestion rate)	200	mg/day	
RfD_{ABS} (dermally-adjusted reference dose)	chemical-specific ^c	mg/kg-day	
AF (skin-soil adherence factor)	0.2	mg/cm ² -event	
ABS_d (dermal absorption factor)	chemical-specific ^b	unitless	
EV (event frequency)	1	event/day	
SA (skin surface area exposed)	2800	cm ²	
^a For non-carcinogens, averaging time equals exposure duration.; ^b See Toxicological Database; ^c See Eq. (12.6).			
$RfD_{ABS} = RfD_o \times ABS_{GI}$			(12.6)
where;			
RfD_{ABS} (dermally-adjusted reference dose)	-	mg/kg-day	
RfD_o (oral reference dose)	chemical-specific ^a	mg/kg-day	
ABS_{GI} (gastro-intestinal absorption factor)	chemical-specific ^a	unitless	
^a See Toxicological Database.			

Inhalation of fugitive particulates

The equations used to calculate the SQSs for inhalation of fugitive particulates pathway are presented in Table 13. Eq. (13.1) is used for exposure to carcinogenic chemicals and Eq. (13.2) is used for exposure to non-carcinogenic chemicals. In both of these equations, particulate

emission factor (*PEF*), which represents an estimate of the relationship between soil contaminant concentrations and the concentration of the contaminants in air as a consequence of particle suspension, is used (US EPA, 2002). *PEF* shows the annual average particulate matter emission resulting from wind erosion. As can be seen from Eq. (13.3), region-

Table 13

Equations used to calculate SQSs for inhalation of fugitive dusts pathway.

$SQS_2^c = \frac{TR \times AT \times 365 \text{ d/yr}}{URF \times EF \times ED \times \left(\frac{1}{PEF}\right)}$			(13.1)
where;			
SQS_2^c (carcinogenic SQS for inhalation of fugitive particulates)	-	mg/kg	
TR (target cancer risk)	10^{-6}	unitless	
AT (averaging time)	70	year	
URF (inhalation unit risk factor)	chemical-specific ^a	$(\text{mg}/\text{m}^3)^{-1}$	
EF (exposure frequency)	350	day/year	
ED (exposure duration)	30	year	
PEF (particulate emission factor)	2.33×10^9 ^b	m^3/kg	
^a See Toxicological Database. ^b See Eq. (13.3) (For residential scenario (source size: 0.01ha) $2.33 \times 10^9 \text{ m}^3/\text{kg}$; for industrial/commercial scenario (source size: 1ha) $7.80 \times 10^8 \text{ m}^3/\text{kg}$).			
$SQS_2^{nc} = \frac{THQ \times AT \times 365 \text{ d/yr}}{EF \times ED \times \left(\frac{1}{RfC} \times \frac{1}{PEF}\right)}$			(13.2)
where;			
SQS_2^{nc} (non-carcinogenic SQS for inhalation of fugitive particulates)	-	mg/kg	
THQ (target hazard quotient)	1	unitless	
AT (averaging time)	30 ^a	year	
EF (exposure frequency)	350	day/year	
ED (exposure duration)	30	year	
RfC (inhalation reference concentration)	chemical-specific ^b	mg/m^3	
PEF (particulate emission factor)	2.33×10^9 ^c	m^3/kg	
^a For non-carcinogens, averaging time equals exposure duration.; ^b See Toxicological Database; ^c See Eq. (13.3) (For residential scenario (source size: 0.01ha) $2.33 \times 10^9 \text{ m}^3/\text{kg}$; for industrial/commercial scenario (source size: 1ha) $7.80 \times 10^8 \text{ m}^3/\text{kg}$).			
$PEF = Q/C \times \frac{3600 \text{ s/h}}{0.036 \times (1-V) \times (U_m/U_t)^2 \times F(x)}$			(13.3)
where;			
PEF (particulate emission factor)	2.33×10^9	m^3/kg	
Q/C (inverse of the mean conc. at the center of a 0.01 ha source)	59.24 ^a	$(\text{g}/\text{m}^2\text{-s})/(\text{kg}/\text{m}^3)$	
V (fraction of continuous vegetative cover)	0.2 (20%)	unitless	
U_m (mean annual wind speed)	3.0	m/s	
U_t (equivalent threshold value of wind speed at 10 m)	8.28	m/s	
F(x) (Function dependent on U_m/U_t derived using Cowherd et al.)	6.67×10^{-2}	unitless	
^a For residential scenario (source size: 0.01 ha) 59.24 g/m ² -s per kg/m ³ ; for industrial/commercial scenario (source size: 1 ha) 19.81 g/m ² -s per kg/m ³ .			

specific parameters such as fraction of vegetative cover (V), mean annual wind speed (U_m) and dispersion factor (Q/C) are used to calculate PEF . Q/C_{wind} is a factor representing the dispersion of fugitive dust emissions in air (Table 9).

Inhalation of volatiles

The equations used to calculate SQSs for inhalation of volatiles are given in Table 14. Eq. (14.1) is used for exposure to carcinogenic chemicals and Eq. (14.2) is used for exposure to non-carcinogenic chemicals. In both of these equations, soil to air volatilization factor (VF), which represents an estimate of the relationship between soil contaminant concentrations and the concentration of the contaminants in air as a consequence of volatilization, is used (US EPA, 2002). As can be seen from Eqs. (14.3) and (14.4), chemical and generic site parameters are used to calculate VF . One of these generic site parameters is Q/C_{vol} , which is estimated by use of air dispersion modeling, representing the dispersion of volatile emissions in ambient air (see Table 9).

Soil saturation concentration: The soil saturation concentration (C_{sat}) corresponds to the contaminant concentration in soil at which the absorptive limits of the soil particles, dissolution limit of soil-water and volatilization limit of soil-air have been reached (US EPA, 1996a). In other words, solid adsorptive surface sites, soil pore water and soil pore air, are saturated with chemical at C_{sat} . Therefore, above this concentration, the soil contaminant may be present in free phase (NAPLs).

The equation and its notation used in calculation of chemical-specific C_{sat} concentrations are given in Table 15. Calculated C_{sat} values must be compared with each volatile inhalation SQS (i.e., carcinogenic and non-carcinogenic SQSs) to assess potential presence of NAPL phase because Henry's law is not applicable when free-phase contaminants are present (US EPA, 1996a). In other words, an accurate VF cannot be estimated

when SQSs for inhalation of volatiles is above C_{sat} . When calculating SQSs for volatile inhalation pathway, C_{sat} values should also be calculated using the same generic soil characteristics (i.e., bulk density, average water content and organic carbon content) used to calculate SQSs (US EPA, 1996a).

For compounds that are liquid at ambient soil temperature, if the volatile inhalation SQS is above C_{sat} , then C_{sat} is set as the SQS for this exposure pathway. Because at C_{sat} the emission flux from soil to air for a chemical reaches an asymptotic value and volatile emissions will not increase above this level, no matter how much more chemical is added to the soil (US EPA, 1996a). This means that there is no volatile inhalation risk for that chemical regardless of the concentration of chemical in soil. However, this situation indicates potential existence of NAPL, which should be considered thoroughly against potential risks to groundwater (US EPA, 1996b). On the other hand, "for organic compounds that are solid at ambient soil temperature, concentrations above C_{sat} do not pose a significant inhalation risk or a potential for NAPL occurrence" (US EPA, 1996b). Owing to this reason, SQSs for this pathway can be neglected.

Migration to groundwater: When deriving the SQSs for migration to groundwater pathway, the potential for leaching of pollutants through soil to an underlying potable aquifer is considered (US EPA, 1996b). The equations used to calculate SQSs for this exposure pathway are given in Tables 16a and 16b. A standard linear equilibrium soil/water partition equation is used to estimate pollutant release into soil leachate (Eq. (16.1)) and to calculate the SQSs. In fact, SQSs are back-calculated from acceptable groundwater concentration which is represented by target soil-leachate concentration, C_w (US EPA, 1996a).

In Turkey, groundwater is commonly used for drinking purposes. Thus, the acceptable groundwater concentration was set according to

Table 14
Equations used to calculate SQSs for inhalation of volatiles pathway.

$SQS_3^c = \frac{TR \times AT \times 365 \text{ d/yr}}{URF \times EF \times ED \times \left(\frac{1}{VF}\right)}$			(14.1)
where;			
SQS_3^c (carcinogenic SQS for inhalation of volatile contaminants)	-	mg/kg	
TR (target cancer risk)	10^{-6}	unitless	
AT (averaging time)	70	year	
URF (inhalation unit risk factor)	chemical-specific ^a	$(\text{mg}/\text{m}^3)^{-1}$	
EF (exposure frequency)	350	day/year	
ED (exposure duration)	30	year	
VF (soil to air volatilization factor)	chemical-specific ^b	m^3/kg	
^a See Toxicological Database; ^b See Eq. (14.3).			
$SQS_3^{nc} = \frac{THQ \times AT \times 365 \text{ d/yr}}{EF \times ED \times \left(\frac{1}{RfC} \times \frac{1}{VF}\right)}$			(14.2)
where;			
SQS_3^{nc} (non-carcinogenic SQS for inhalation of volatile contaminants)	-	mg/kg	
THQ (target hazard quotient)	1	unitless	
AT (averaging time)	30 ^a	year	
EF (exposure frequency)	350	day/year	
ED (exposure duration)	30	year	
RfC (inhalation reference concentration)	chemical-specific ^b	mg/m^3	
VF (soil to air volatilization factor)	chemical-specific ^c	m^3/kg	
^a For non-carcinogens, averaging time equals exposure duration.; ^b See Toxicological Database; ^c See Eq. (14.3).			
$VF = \frac{Q/C \times (3.14 \times D_A \times T)^{1/2} \times (10^{-4} \text{ m}^2/\text{cm}^2)}{(2 \times \rho_b \times D_A)}$			(14.3)
$D_A = \frac{[(10^{10}/3) D_i H' + 10^{10}/3 D_w/n^2]}{\rho_b K_d + 10^{10}/3 H'}$			(14.4)
where;			
VF (soil to air volatilization factor)	chemical-specific	m^3/kg	
D_A (apparent diffusivity)	chemical-specific	cm^2/s	
Q/C (inverse of the mean conc. at the center of a 0.01 ha source)	27.61 ^a	$(\text{g}/\text{m}^2\text{-s})/(\text{kg}/\text{m}^3)$	
T (exposure interval)	9.5×10^8	s	
ρ_b (dry soil bulk density)	1.5	g/cm^3	
θ_a (air filled soil porosity)	$n - \theta_w$	cm^3/cm^3	
n (total soil porosity)	$1 - (\rho_b/\rho_s)$	cm^3/cm^3	
θ_w (water filled soil porosity)	0.15	cm^3/cm^3	
ρ_s (soil particle density)	2.65	g/cm^3	
D_i (diffusivity in air)	chemical-specific ^b	cm^2/s	
H' (Henry's law constant)	chemical-specific ^b	unitless	
D_w (diffusivity in water)	chemical-specific ^b	cm^2/s	
K_d (soil-water partition coefficient)	chemical-specific ^{b,c}	cm^3/g	
K_{oc} (soil organic carbon partition coefficient)	chemical-specific ^b	cm^3/g	
f_{oc} (fraction of organic carbon in soil)	0.006	g/g	
^a For residential scenario (source size: 0.01ha) 27.61 $\text{g}/\text{m}^2\text{-s}$ per kg/m^3 ; for industrial/commercial scenario (source size: 1ha) 8.96 $\text{g}/\text{m}^2\text{-s}$ per kg/m^3 .			
^b See Chemical Database; ^c For organics $K_d = K_{oc} \times f_{oc}$, for metals K_d value is used.			

Table 15
Equation used to calculate soil saturation concentration.

$C_{sat} = \frac{S}{\rho_b} [K_d \rho_b + \theta_w + H' \theta_a]$			(15.1)
where;			
C_{sat} (soil saturation concentration)	-	mg/kg	
S (solubility in water)	chemical-specific ^a	mg/L	
ρ_b (dry soil bulk density)	1.5	kg/L	
K_d (soil-water partition coefficient)	chemical-specific ^{a,b}	L/kg	
K_{oc} (soil organic carbon partition coefficient)	chemical-specific ^a	L/kg	
f_{oc} (fraction of organic carbon in soil)	0.006 (%0.6)	g/g	
θ_w (water filled soil porosity)	0.15	cm^3/cm^3	
H' (Henry's law constant)	chemical-specific ^a	unitless	
θ_a (air filled soil porosity)	$n - \theta_w$	cm^3/cm^3	
n (total soil porosity)	$1 - (\rho_b/\rho_s)$	cm^3/cm^3	
ρ_s (soil particle density)	2.65	g/cm^3	
^a See Chemical Database; ^b For organics $K_d = K_{oc} \times f_{oc}$, for metals K_d value is used.			

the standards of Turkish Standardization Institute, TS-266 Water Intended for Human Consumption Standards (TSE, 2005). For the compounds that are not included in TS-266, drinking water standards of WHO (WHO, 2008) were used. For the compounds that are not covered by any of these standards, health based limits (HBL), which are risk based drinking water concentrations, were calculated and used. The equations used to calculate carcinogenic and non-carcinogenic HBLs are presented in Eqs. (16.2) and (16.4), respectively. While calculating car-

cinogenic HBL, age-adjusted drinking water ingestion rate is included (Eq. (16.3)).

The dilution factor represents the reduction in soil leachate pollutant concentrations by mixing in the aquifer, expressed as the ratio of leachate concentration to the concentration in groundwater at the receptor point, e.g., drinking water well (US EPA, 1996a). As a conservative approach, the equations used for development of DF does not account for attenuation (e.g., adsorption and degradation) of

Table 16a

Equations used to calculate SQSs for migration to groundwater pathway.

$SQS_4 = C_w (K_d + \frac{\theta_w + \theta_a H'}{\rho_b})$			(16.1)
where;			
SQS_4 (SQS for migration to groundwater pathway)	-	mg/kg	
C_w (target soil leachate concentration)	chemical-specific ^a	mg/L	
K_d (soil-water partition coefficient)	chemical-specific ^{b,c}	L/kg	
K_{oc} (soil organic carbon partition coefficient)	chemical-specific ^b	L/kg	
f_{oc} (fraction of organic carbon in soil)	0.002 (% 0.2)	g/g	
θ_w (water filled soil porosity)	0.3	cm ³ /cm ³	
θ_a (air filled soil porosity)	$n - \theta_w$	cm ³ /cm ³	
H' (Henry's law constant)	chemical-specific ^{b,d}	unitless	
n (total soil porosity)	$1 - (\rho_b/\rho_s)$	cm ³ /cm ³	
ρ_b (dry soil bulk density)	1.5	kg/L	
ρ_s (soil particle density)	2.65	kg/L	
^a $C_w = DF \times (TS-266, WHO \text{ or } HBL \text{ standards})$; ^b See Chemical Database.			
^c For organics $K_d = K_{oc} \times f_{oc}$; for metals K_d value is used.			
^d Assumed to be zero for inorganic contaminants except mercury.			
$HBL^c (\text{mg/L}) = \frac{TR \times AT \times 365 \text{ d/yr}}{EF \times SF_o \times IF_{w-adj}}$			(16.2) ^a
where;			
HBL^c (carcinogenic health based limit)	-	mg/L	
TR (target cancer risk)	10^{-6}	unitless	
AT (averaging time)	70	year	
EF (exposure frequency)	350	day/year	
SF_o (oral slope factor)	chemical-specific ^b	(mg/kg-day) ⁻¹	
IF_{w-adj} (age-adjusted drinking water ingestion rate)	1.086 ^c	L-year/kg-day	
^a This equation is adopted from URL 5; ^b See Toxicological Database; ^c See Eq. (16.3)			
$IF_{w-adj} = \frac{ED_{1-6} \times IR_{w/1-6}}{BW_{1-6}} + \frac{ED_{7-31} \times IR_{w/7-31}}{BW_{7-31}}$			(16.3) ^a
where;			
IF_{w-adj} (age-adjusted drinking water ingestion rate)	1.086	L-year/kg-day	
$IR_{w/1-6}$ (drinking water ingestion rate - child)	1	mg/day	
$IR_{w/7-31}$ (drinking water ingestion rate - adult)	2	mg/day	
ED_{1-6} (exposure duration during ages 1-6)	6	year	
ED_{7-31} (exposure duration during ages 7-31)	24	year	
BW_{1-6} (average body weight from ages 1-6)	15	kg	
BW_{7-31} (average body weight from ages 7-31)	70	kg	
^a This equation is adopted from URL 5.			

Table 16b

Equations used to calculate SQSs for migration to groundwater pathway (cont'd).

$HBL^{nc} (\text{mg/L}) = \frac{THQ \times AT \times BW \times 365 \text{ d/yr}}{EF \times ED \times \frac{1}{RfD_o} \times IR_w}$			(16.4) ^a
where;			
HBL^{nc} (non-carcinogenic health based limit)	-	mg/L	
THQ (target hazard quotient)	1	unitless	
AT (averaging time)	30 ^b	year	
BW (body weight)	70	kg	
EF (exposure frequency)	350	day/year	
ED (exposure duration)	30	year	
RfD_o (oral reference dose)	chemical-specific ^c	mg/kg-day	
IR_w (drinking water ingestion rate)	2	l/day	
^a This equation is adopted from URL5.			
^b For non-carcinogens, averaging time equals exposure duration.; ^c See Toxicological Database.			
$DF = 1 + \frac{Kid}{IL}$			(16.5)
where;			
DF (dilution factor)	-	unitless	
K (aquifer hydraulic conductivity)	site-specific	m/year	
i (hydraulic gradient)	site-specific	m/m	
I (infiltration rate)	site-specific	m/year	
d (mixing zone depth) ^a	site-specific	m	
L (source length parallel to ground water flow)	site-specific	m	
^a See Eq. (16.6)			
$d = (0.0112L^2)^{0.5} + d_a \{1 - \exp[(-LI)/(Kid_a)]\}$			(16.6)
where;			
d (mixing zone depth)	-	m	
L (source length parallel to ground water flow)	site-specific	m	
I (infiltration rate)	site-specific	m/year	
K (aquifer hydraulic conductivity)	site-specific	m/year	
d_a (aquifer thickness)	site-specific	m	

contaminants. DF can be determined by use of groundwater simulation models (US EPA, 1996a) or use of a simple water-balance equation (as given in Eqs. (16.5) and (16.6)). A detailed discussion of the general approach to determination of DF and the related studies

are presented by the US EPA (1996a). However, due to insufficient field data, a generic DF could not be estimated for Turkey. Instead, the generic DF of 10 was used, which is an over-conservative and at the same time a reasonable value compared to other countries'

generic DF values. For example, the Netherlands and Norway use a generic DF of 10; Canada 50; and the US EPA 20 for a source size of 0.2 ha and 0.10 for a source size of 12 ha. As a result, SQSs are calculated for two DF values, 1 and 10. It is decided to use a DF of 1, in case the depth to aquifer is less than 3 m, or aquifer is fractured or karstic; or source area is greater than or equal to 10 ha; in all other conditions DF is accepted as 10 (Ünlü et al., 2009).

By multiplying the acceptable groundwater concentration (TS-266, WHO or HBL standards) by the DF, a target leachate concentration, C_w is obtained. If DF is taken as 10, for an acceptable groundwater concentration of 0.05 mg/L, the target soil leachate concentration will be found as $0.05 \times 10 = 0.5$ mg/L (US EPA, 1996a). If DF assumed to be 1, it means the worst case scenario applies and no dilution or attenuation between the source and the receptor well is expected. Thus, target soil leachate concentration equals to the acceptable groundwater concentration.

To summarize, soil contaminant concentration, SQS in mg/kg, is calculated by use of the corresponding target soil-leachate concentration, C_w in mg/L (Eq. (16.1)). In the end, the SQSs calculated are compared with C_{sat} , to check for the existence of free phase substance. If the SQS for migration to groundwater pathway is higher than C_{sat} , then C_{sat} is set as the SQS for this exposure pathway.

2.3.2. Soil Quality Standards for special case chemicals

SQSs for most of the chemicals can be derived readily by using the equations given in Section 2.3.1; however, for some chemicals particular attention is needed (URL 4, 2011). In the following sections, the chemicals that need further attention in derivation of SQSs are discussed. These chemicals are cadmium, chromium, lead, PCBs, and dioxins.

Cadmium: For cadmium, two different RfD_o values are presented by IRIS; one of them is based on cadmium intake by water (0.0005 mg/kg-day) and the other is based on the intake by food (0.001 mg/kg-day) (URL 3, 2011; URL 5, 2011). Since RfD_o values differ with respect to the exposure type, additional care should be taken for risk assessment of cadmium depending on the purpose of use. Since exposure to cadmium by groundwater ingestion is covered by TS-266 standards (0.005mg/L) and the SQS for this pathway is calculated based on this criterion, RfD_o value for dietary exposure was used for derivation of the generic SQSs for direct ingestion.

Chromium: Although SQSs for ingestion are more conservative than most of the generic SQSs for inhalation of fugitive particulates, it is not the case for chromium (US EPA, 1996b). Because of the carcinogenicity of hexavalent chromium (Cr^{+6}) through inhalation exposure, SQS calculated for this pathway is lower than the SQS for ingestion. Therefore, due attention should be paid for estimating site-specific SQSs, especially when site conditions are convenient for significant dust emissions; e.g., dry, dusty soils, high average annual wind speeds, vegetative cover less than 50 percent (US EPA, 1996b).

Since different valences of chromium produce different toxicities (US EPA, 1996b), use of valent-specific data is recommended for the sites that are likely to be contaminated with chromium (URL 5, 2011). Because of the high carcinogenic potency of Cr^{+6} , chromium (total) is based on the SQSs calculated for Cr^{+6} .

Lead: Lead is considered to be a special case chemical by US EPA, because of the difficulty in developing a RfD_o (URL 5, 2011). For this reason, a generic soil screening level for lead has not been calculated by US EPA. However, models are recommended in order to assess lead exposure. One of these models is *Integrated Exposure Uptake Biokinetic Model (IEUBK)*, which is designed specifically for evaluating lead exposures in children (US EPA, 1994). With use of this model, US EPA (1994) has calculated a Soil Screening Level (SSL) of 400 ppm for residential land-use. Another model developed by US EPA is *Adult Lead Model (ALM)*. This model is used for assessing risks associated with non-residential adult exposures to lead in soil (US EPA, 1999, 2003b). ALM has also been used by the DEFRA (2002) and soil quality standard was determined as 450 mg/kg, which is close to the value determined by US EPA. Both IEUBK and ALM models are based on a different calculation methodol-

ogy; and they also use different toxicity criteria for calculations, such as biokinetic slope factor in $\mu\text{g}/\text{dL}$ per $\mu\text{g}/\text{day}$ (US EPA, 2003b).

US EPA (URL 5, 2011) recommends 400 mg/kg for residential soils for screening purposes, and proposes 15 $\mu\text{g}/\text{L}$ as maximum concentration level in water (US EPA's action level for water). In this regard, since no toxicity criteria are available for lead, 400 mg/kg is adopted as the generic SQS for ingestion-dermal absorption pathway. For migration to groundwater pathway, the acceptable groundwater concentration given in TS-266 (10 $\mu\text{g}/\text{L}$) is used for calculations of the Turkish SQS.

Polychlorinated biphenyls (PCBs): PCBs refer to a group of chemicals, which contain 209 individual compounds (congeners), and most often the trade name Aroclor is used to describe PCBs (URL 7, 2011). Aroclors are coded with respect to their parent molecule (the first two digits) and their chlorine content by weight (last two digits) (URL 5, 2011). For example, the parent molecule for Aroclor 1260 is biphenyl and it contains 60% chlorine.

PCBs are classified among probable human carcinogens. Since most toxicity testing has been done on these specific commercial mixtures i.e., Aroclors (URL 7, 2011), PCB exposures are often characterized, as in France, in terms of Aroclors (URL 5, 2011). However, US EPA founds this approach imprecise and inappropriate (URL 5, 2011) because, once these mixtures are released to environment, they differ in composition due to partitioning, biotransformation, and bioaccumulation (URL8). Therefore, congener or isomer or total PCBs analyses are recommended by US EPA (URL 5, 2011).

In IRIS (URL 6, 2011), different toxicological data are proposed with respect to the following groups of PCBs: (i) high risk and persistence, (ii) low risk and persistence, and (iii) lowest risk and persistence. IRIS defines Aroclor 1260 as a persistent mixture creating more tumors than less persistent mixture Aroclor 1016 (URL 6, 2011). On this account, Aroclor 1016 has been considered to pose low risk, whereas all other Aroclors have been considered to pose high risk toxicity values and appropriate values has been assigned (URL 5, 2011). In this respect, generic SQSs were derived based on low risk group PCBs (including only Aroclor 1016 mixtures) and high risk group PCBs (including mixtures other than Aroclor 1016). For this purpose, the corresponding toxicity criteria given in IRIS (URL 6, 2011) were used.

As a result, ingestion SQS for high risk group PCBs is found as 0.2 mg/kg, and for low risk group it is found as 1 mg/kg for residential land use. For commercial/industrial land-use outdoor worker, 0.8 mg/kg and 4 mg/kg are found for high risk and low risk group PCBs, respectively. US EPA (1990) recommends 1 ppm for residential land-use and 10 to 25 ppm for industrial land-use (regardless of the pathways) as the *soil action levels*. These values are also calculated by US EPA with respect to 10^{-6} risk level. Considering the purpose of use of SQSs and soil action levels of US EPA, Turkish SQSs derived are compatible with these values.

Dioxins: Dioxins represent a group of chlorinated organic chemicals that have similar structures. Although these chemicals have similar toxicological properties, their degree of toxicity differs (URL 5, 2011). In order to adjust the measured concentration to a toxicity equivalent (TEQ) concentration (i.e., to calculate overall toxicity of the dioxin mixture), toxicity equivalence factors (TEFs) are used. Dioxin-like TEFs are provided for dioxins, furans and PCBs (URL 5, 2011).

The isomer 2,3,7,8 TCDD is defined as the most widely studied compound in this class. This compound represents the reference compound for this class (i.e. TEF = 1.0), thus it is simply called as "dioxin"; it is often used to refer to the complex mixtures of 2,3,7,8 TCDD and related compounds (US EPA, 2000). Thus, SQSs are calculated for 2,3,7,8 TCDD by using the toxicological data given in RAIS (URL 3, 2011). Ingestion SQS is calculated as 0.000004 mg/kg (0.004 ppb) for residential land use and 0.00002 mg/kg (0.02 ppb) for commercial/industrial land use - outdoor worker. Whereas, US EPA (1998) recommends *soil action levels* of 1 ppb for residential land-use (which corresponds to 2.5×10^{-4} lifetime cancer risk at residential exposure) and 5-20 ppb for commercial/industrial land-use (5 ppb corresponds to 1.3×10^{-4} lifetime can-

cer risk at commercial/industrial exposure) regardless of the pathways. Considering the purpose of use of Turkish SQSs (i.e., to screen sites that do not need further assessment) and the target risk level (10^{-6}) used for calculation of ingestion SQSs, the calculated SQSs are found to be compatible with the US EPA's values.

3. Results and discussions

For development of SQSs, a comprehensive literature review through available documents, reports, regulatory standards and guidelines of the European Countries, the US EPA and Canada was conducted to grasp the approaches and the procedures utilized for derivation of human health risk based SQSs. Reviewing the SQSs of different countries, the important results can be summarized as follows. All industrialized countries have been facing with severe land contamination problems forcing them to configure their SQSs in compliance with the physical, political and economical characteristics of their country. The general approach of these countries (mostly EU and North American countries) is to set sustainable standards. Studies show that, although risk based SQSs have already been defined by many countries, the research in this field continue for updating these values. Although adopting the human health risk based SQSs derived by one of these countries were standing as an alternative to developing national SQSs, differences in SQSs (mentioned in Section 2.1.1) demonstrated the significance of the region-specific characteristics and the needs of the country regarding development of national SQSs. As a result, the experiences and findings of these countries were taken into account. The approaches and procedures of other countries established with expertise have been investigated thoroughly and the key components of the study for derivation of Turkish human health risk based SQSs were determined.

Both in Turkey and Europe, the common industries that can cause soil contamination and the priority soil contaminants were identified. These chemicals, as well as the list of chemicals that take place in the regulations of other countries, formed a bases for identifying the priority soil contaminants for Turkey.

For derivation of human health risk based SQSs for Turkey, the approach of US EPA, which is based on the same principles as that of the other countries, was adopted as an appropriate way of fulfilling the needs of the regulation. Consequently, the US EPA's methodology was implemented, and the approaches of the other countries (especially, the Netherlands, Norway and the UK) were utilized (sometimes as a guide, and sometimes for comparison) for development of Turkish SQSs.

The target risk level (10^{-6}) was identified according to the decision of the TMOEUA by considering the intended use of Turkish SQSs. The exposure scenarios (i.e. residential scenario, and commercial/industrial scenario for indoor workers and outdoor workers) and the exposure pathways (i.e. soil ingestion-dermal contact, inhalation of fugitive dusts and volatiles, migration to groundwater) to be used for development of generic SQSs for Turkey were identified. The exposure parameters and contact rates applicable for these scenarios and pathways were determined. The generic site characteristics for Turkey to be used in calculation of SQSs were specified. The physical-chemical and toxicological data required for calculation of SQSs were identified and the necessary data were compiled in an MS Excel based data library. Finally, Turkish SQSs were calculated for three land-use scenarios (i.e. residential land use, commercial/industrial land use: outdoor worker and commercial/industrial land use: indoor worker) and four primary exposure pathways (i.e. direct soil ingestion-dermal contact, inhalation of volatiles, inhalation of fugitive particulates and migration to groundwater).

3.1. The use of generic Soil Quality Standards

The generic SQSs represent the soil concentrations of contaminants, which are calculated assuming the reasonable maximum exposure of human receptors to contaminants in a (current or future) residential land use. In this respect, the generic SQSs will be used to screen the sites

that do not need further attention and to identify the sites that need further investigation (i.e., site-specific risk assessment). In addition to these aims, generic SQSs could also be utilized for determination of the initial clean-up goals when site-specific data are lacking (URL 5, 2011).

A systematic approach for management of the contaminated sites was required by the new Turkish soil pollution control regulation (TMOEUA, 2010). In the framework of contaminated sites management system of this regulation, generic SQSs take part in the generic risk assessment phase. Generic risk assessment involves comparison of the potential exposure pathways with the generic exposure pathways, both of which are defined by the generic Conceptual Site Model, CSM. Site concentrations can be compared with the generic SQSs, for which the actual exposure pathways are compatible with the generic exposure pathways. However, surface soil concentrations must be compared with the generic SQSs derived for ingestion-dermal contact and inhalation of fugitive particulates pathways, whereas subsurface soil concentrations must be compared with the generic SQSs derived for inhalation of volatiles and migration to groundwater pathways.

Sites where the measured surface and subsurface soil concentrations below the generic SQSs can be screened out; that is, such sites are clean and need no further investigation. The generic SQSs for the pathways that are not present at the actual site conditions can be disregarded. On the other hand, the other exposure pathways, which are present at the actual site but not considered in the calculation of generic SQSs should be assessed under site-specific risk assessment phase.

3.2. Special considerations for generic Soil Quality Standards

Calculation of SQSs require special consideration for background soil concentrations, detection limits of chemical measurements, presence of multiple contaminants, and presence of free phase contaminants in soil. Each of these issues are discussed separately in following subsections.

3.2.1. Background soil concentrations

While comparing the site concentrations with the generic SQSs, the background soil concentrations should also be considered. US EPA (URL 5, 2011) defines two types background concentrations; (i) *natural background concentration* (usually limited to metals), and (ii) *anthropogenic background concentration* (includes both organic and inorganic contaminants).

For some chemicals, the SQSs derived by using risk-based approaches may be lower than the background soil concentrations. US EPA (URL 5, 2011) states that arsenic, aluminum, iron and manganese are among the chemicals for which background soil concentrations may exceed generic SQSs. Therefore, not only the potentially contaminated site, but also its surroundings should be assessed in order to identify the typical background concentrations for the site. If generic SQSs for the metal contaminants are below the background soil concentration, then background soil concentration may be accepted as the soil quality criterion for that chemical (SFT, 1999).

3.2.2. Detection limit for chemicals

As for background soil concentrations, for each priority soil contaminants listed in Table 2, technically feasible detection limit should be considered during generic risk assessment studies. If calculated generic SQS is below this detection limit, then soil quality criteria should be adjusted to the detection limit.

3.2.3. Sites with multiple contaminants

SQSs are developed based on chemical-specific toxicity criteria to eliminate the sites that do not need further investigation. However, there might be sites that are contaminated with multiple chemicals. In such circumstances, generic risk assessment must be performed for each contaminant found at the site (TMOEUA, 2010). In other words, the site with multiple contaminants must be assessed with respect to

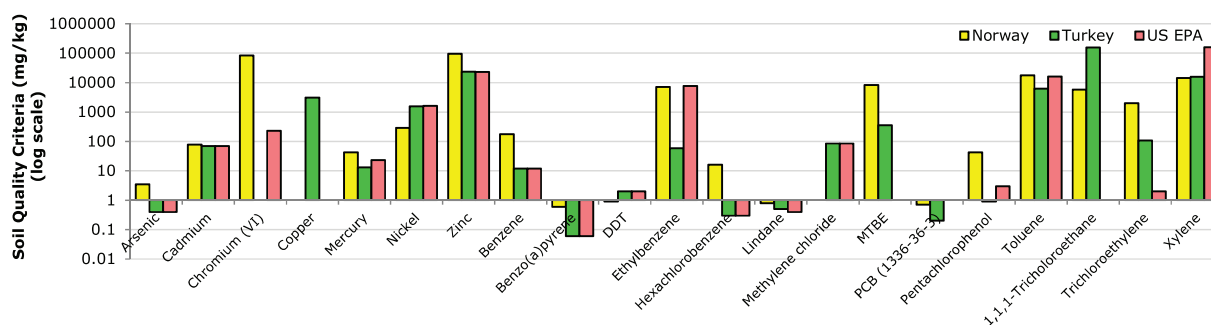


Fig. 2. Comparison of Norwegian, Turkish and US EPA SQSs for ingestion of soil and dermal contact pathway.

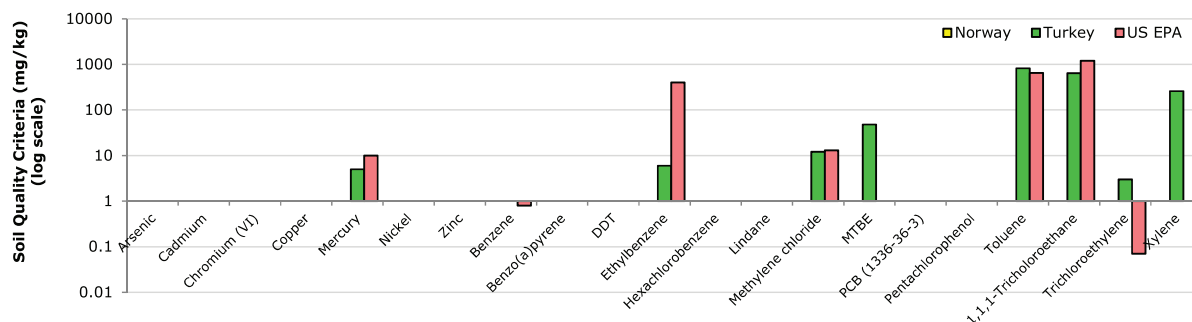


Fig. 3. Comparison of Norwegian, Turkish and US EPA SQSs for inhalation of volatiles pathway.

the generic SQSs of each contaminant. During the assessment, the target organ/system under threat should also be considered. Since different chemicals might affect the same target organ/system, the human health effects of each chemical should be evaluated before screening out any site (URL 5, 2011). For this purpose, information about chemical specific cancer classification and impacted Target Organ/System are needed. Such information was compiled from the RAIS database (URL 3, 2011) and used for reviewing the target organ/system of the contaminants at site to consider the cumulative impact of multiple contaminants on human health.

3.2.4. Presence of free-phase contaminants

As described in Section 2.3.1 “Soil Saturation Concentration”, if calculated SQS for a chemical is higher than C_{sat} , then C_{sat} is set as the SQS for that chemical. If site concentrations exceed C_{sat} , it indicates a potential presence of free NAPL in soil, which poses risk to groundwater (US EPA, 1996a). Therefore, in such circumstances, further site assessment is required. However, it should also be noted that free-phase contaminants may also be present at concentrations below C_{sat} if multiple organic contaminants are present at site (US EPA, 1996b).

3.3. Comparison of Turkish SQSs with soil criteria of other countries

As discussed in Section 2.1.1, the approaches used for derivation of SQSs differ from country to country, which results in 10 to 100 folds of differences in SQSs. Since soil quality criteria in these countries are based on different assumptions and used for different purposes, it is very difficult to compare the SQSs of different countries with each other due to limited common ground. Therefore, a thorough understanding of the procedure followed by each country for derivation of soil quality criteria is needed to compare them meaningfully. Otherwise, the differences in SQSs of different countries can be misinterpreted.

In this context, to compare the developed Turkish SQSs with the SQSs of other countries, the following factors were taken into consideration: (i) purpose of use; i.e., the role of SQSs in the contaminated sites management system (e.g., screening), (ii) the exposure scenario used (e.g., residential land use), (iii) the potential receptors considered (e.g., hu-

man beings), (iv) the pathways of concern (e.g., ingestion-dermal contact, inhalation of volatiles, inhalation of fugitive dusts, migration to groundwater), and (v) target risk level (e.g., 10^{-6}).

Among the countries reviewed in this study, other than US EPA's soil screening levels (SSLs), the SQSs of Norway were found to be comparable with the Turkish SQSs. Norwegian generic soil quality values also address the intermediate risk level and used for determination of the need for further investigation. The Norwegian SQSs are based on the sensitive land use, in which ecological receptors are considered as well as human receptors. However, human health risk based Norwegian SQSs are presented solely with respect to the pathways considered (SFT, 1999). Overall, in the Norwegian approach for derivation of SQSs, pathway specific standards are calculated and the values for all pathways are combined by taking the harmonic mean to produce a total human exposure limit. This value is then compared with the ecological risk limit and the lowest of these two values is considered as the SQS, which is then adjusted with respect to the detection limit and background soil concentrations.

Pathway specific comparisons of Turkish SQSs, US EPA's soil screening levels (SSLs) and the Norwegian SQSs for a number of metals and organic compounds that are important for soil pollution were presented in Figs. 2–5. While comparing Turkish SQSs with the soil quality criteria of Norway, it should be kept in mind that Norwegian soil criteria are developed based on 10^{-5} target risk level. Despite the difference in target risk levels, Norwegian soil criteria are comparable with Turkish SQSs for ingestion-dermal contact pathway for some substances, such as cadmium, mercury, DDT, lindane, PCBs, xylene. However, significant differences appear for some other substances, such as chromium (VI), copper, zinc, ethylbenzene, and 1,1,1-trichloroethane (Fig. 2). Similar observations can also be made for inhalation of fugitive dusts and migration to groundwater pathways (Figs. 4 and 5). However, Norwegian SQSs are not comparable both with US EPA SSLs and Turkish SQSs for inhalation of volatiles pathway (Fig. 3).

A remarkable fact is that, even though Norwegian SQSs are based on 10^{-5} target risk level and Turkish SQSs are based on 10^{-6} target risk level, Norwegian SQS values for some chemicals are lower than Turkish SQSs. The reason for this situation may be explained partly by

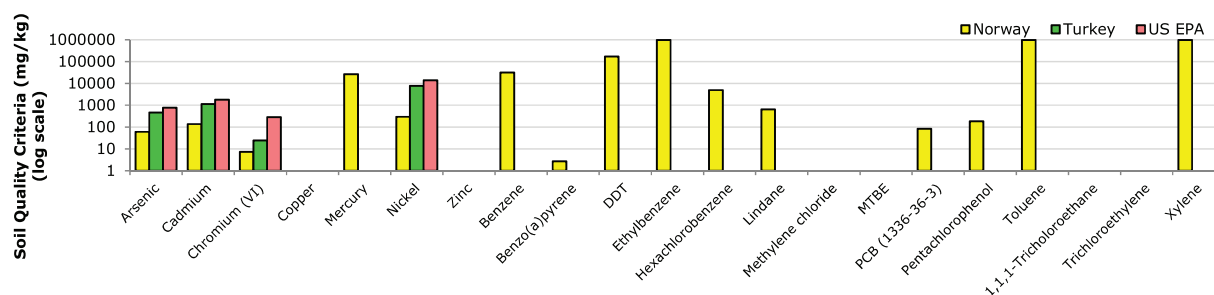


Fig. 4. Comparison of Norwegian, Turkish and US EPA SQSs for inhalation of fugitive dust pathway.

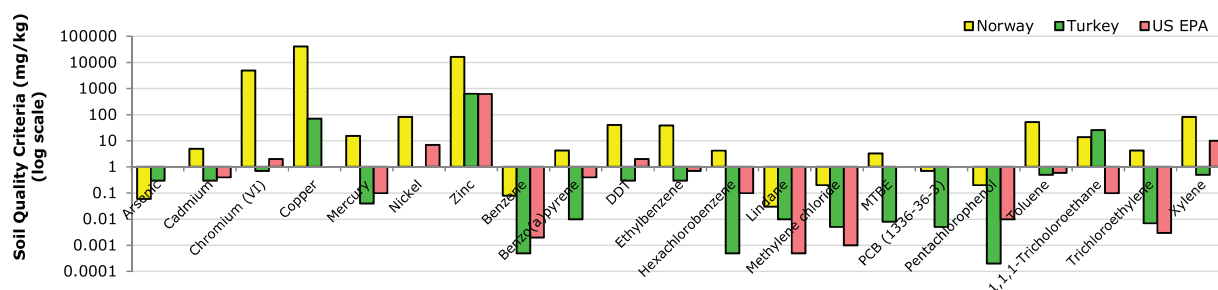


Fig. 5. Comparison of Norwegian, Turkish and US EPA SQSs for migration to groundwater pathway.

the use of different toxicological data since Norwegian soil guideline values (SFT, 1999) are derived using the toxicological data available at that time. This difference, however, can also be explained mainly by the differences in approaches, assumptions, parameters and generic site characteristics defined.

As can be seen from Figs. 2–5, Turkish SQSs are reasonably comparable with US EPA's SSLs, since Turkish SQSs were derived by adopting the US EPA's methodology. Especially, when the ingestion-dermal contact pathway is considered, it can be seen that SQSs derived are close to the SSL of US EPA. This is because the exposure parameters and the contact rates were assumed to be applicable to Turkey. On the contrary, the differences in soil quality criteria are due to the use of country-specific values for air dispersion factor (Q/C), mean annual wind speed (U_m), fraction of vegetative cover (V), and acceptable groundwater concentration (C_w) and dilution factor (DF).

4. Conclusions

Soil clean-up brings serious economical burden to governments and the public. Because of economical and technological drawbacks, soil clean-up to background levels is regarded as infeasible in most of the cases. Moreover, the available labor, time, equipment and financial resources to be allocated for remediation of contaminated sites have to be optimized, since it is not possible to overcome all cases simultaneously. Therefore, sites needing remediation should be determined with respect to the risk they pose. An adequate soil policy with approaches balancing the threats on human health and environment with the efforts and funding utilized for remediation are implemented in most of the countries who made a considerable progress in managing contaminated soils.

Soil policy in Turkey has evolved over the last decade from a rigid soil quality assessment procedure bringing in impracticable and strict remediation measures into a more flexible, systematic and comprehensive management system adopting the risk-based Soil Quality Standards as the main instrument for soil quality assessment. The derivation of human health risk based SQSs involves use of a large amount of information and data, which are progressively subject to changes or upgrades. SQSs may be sensitive to these changes, thus, the derivation process is very dynamic in nature. SQSs are sensitive to changes in exposure parameters and contact rates that differ with respect to the exposure

scenario considered and the receptors of concern; generic site characteristics with respect to soil, hydrogeologic and climatic conditions; and physical-chemical and toxicological properties of contaminants, which are periodically upgraded or newly produced through recent scientific research. Today, in Turkey more attention should be paid to maintain the sustainability of SQSs by continuous monitoring and supporting the improvements in the toxicological and other scientific research and upgrading the SQSs with the use of latest information or data produced. Thus, conceptual framework should be enhanced with the additional scientific and technical infrastructure that ensures the sustainability of the Turkish contaminated sites management system.

Based on the proposed conceptual framework and findings of this study, the following points are highlighted for researchers planning for the development of health-risk-based Soil Quality Standards:

- A priori knowledge of all soil polluting activities in the country and the relevant chemicals that will potentially be released into the soil environment from these activities are essential to ease the decision making in case of pollution events.
- The generic site characteristics that are representative of the country-specific soil, meteorological and hydrogeological properties and conservative for a range of site conditions need to be specified.
- Derivation of “air dispersion factors” for fugitive dusts and for volatiles needed for derivation of SQS for inhalation pathway requires specific meteorological data and intensive air quality modeling runs.
- Similarly, derivation of “dilution factor” needed for derivation of SQS for groundwater pathway requires specific hydrogeological data and intensive groundwater transport modeling runs. The most stringent SQSs are obtained for groundwater pathway, thus determining the proper dilution factor is a challenge and plays a critical role.
- Decision on the “target risk level” is one the most challenging and sensitive political issues since it affects the numerical values of calculated SQSs the most.
- Cadmium, chromium, lead, PCBs, and dioxins requires particular attention for derivation of their SQSs. More recently, similar special attention is applicable for so called, “micro and emerging pollutants”.
- A priori specification of the purpose of use of SQSs (i.e., the role of SQSs in the contaminated sites management system of the country) is

essential for their implementation. In this regard, the generic SQSS can be used to screen the sites that do not need further attention and to identify the sites that need site-specific risk assessment. In addition, generic SQSSs could also be used for setting up the initial clean-up goals when site-specific data are lacking.

- Calculation of SQSSs require special consideration for background soil concentrations, detection limits of chemical measurements, presence of multiple contaminants, and presence of free phase contaminants in soil.
- Despite the employment of common underlying principles for the development of health-risk-based soil quality criteria, 10 to 100 folds of difference in SQSSs for the same contaminant can be observed due to use of different assumptions and different parameter values in calculations by different countries. Therefore, for meaningful comparisons of SQSSs, a thorough understanding of the derivation procedure followed by each country is essential.
- Integration of ecological risks into soil SQSSs may be quite useful for a wide range of soil pollution cases; however, it is a big challenge for most countries because ecological risk assessment requires extensive data and knowledge on exposure and dose-response assessments performed on ecological receptors, which may be produced only through well-planned national research programs.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

CRedit authorship contribution statement

Meltem İpek: Methodology, Investigation, Data curation.
Kahraman Ünlü: Conceptualization, Formal analysis, Writing - original draft, Supervision.

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