Swedish University of Agricultural Sciences
Department of Soil and Environment

Studies of the transfer of radionuclides and dose rate assessments in the pasture lands located in the counties of Uppsala and Jämtland

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Master's Thesis in Environmental Science
Environmental Pollution and Risk Assessment - Master's Programme

Institutionen för mark och miljö, SLU
Examensarbeten 2012:06

Uppsala 2012

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EX0431, Independent Project in Environmental Science - Master´s thesis, 30 credits, A2E
Environmental Pollution and Risk Assessment - Master's Programme 120 credits

Institutionen för mark och miljö, SLU, Examensarbeten 2012:06
Uppsala 2012

Keywords: radionuclides, dose rate, transfer factor, migration, Chernobyl accident

Online publication: http://stud.epsilon.slu.se

Abstract

The study was conducted in the pastoral lands of two Counties, Uppsala and Jämtland Counties. Soil and grass samples were collected from six study areas. Two from Uppsala County: Lövstalöt and Möjsjövik, and four from Jämtland County: Backfors, Hallen, Myrviken and Vikdrolet. Milk samples were also collected from Lövstalöt (Uppsala County) and Hallen and Oviken (Jämtland County). The study aims at analyzing the presence and activity concentration of natural radionuclides and ^{137}Cs from Chernobyl accident and determines their transfer factor from soil to grass and from grass to milk. It also describes the migration of ^{137}Cs and the homogeneity of concerned natural radionuclides in the soil profile of the study areas. The third objective of the study was to assess in situ and laboratory outdoor dose rate in air of radionuclides in the study areas and make comparison between Counties and the methods. Higher concentrations of natural radionuclides were found in Jämtland study areas than in Uppsala due to the geological background of the area rich in natural radionuclides. But, higher concentration of ^{137}Cs was found in Uppsala than in Jämtland due to higher anthropogenic deposition from Chernobyl accident. Activity concentrations of the radionuclides were higher than suggested by UNSCEAR. In Uppsala study areas, higher total annual effective dose rate of radionuclides were found in the laboratory measurement than in situ. A similar finding was obtained in Jämtland study areas except in Hallen. In Jämtland County, the contribution of artificial radionuclide (^{137}Cs) was not very significant to the total annual effective dose rate and much concern could be given to the natural radionuclides dose rate assessment. In Uppsala County, however, it can be advocated that equal concerns should be given to both artificial radionuclide (^{137}Cs) and natural radionuclide for dose rate assessment due to their significant contribution to the total annual effective dose rate estimated in the study.

Acknowledgement

I would like to thank my supervisor Jose Luis Gutierrez Villanueva for his training, guidance and invaluable supervision and comments for successfully completion of this project work. My deep gratitude also extends to Klas Rosén not only for organizing Radioecology course, comments and encouragement but also for being so helpful in personal matters in many ways. I further acknowledge my appreciation to Radioecology group especially Mykhailo Vinichuk, Joris Van Schaik and Stefan Bengtsson during samples measurement.

Taking this opportunity, I would like to express my deep appreciation and admiration to my elder brothers Mogos Afewerki, Tekeste Negash, Ficre Zehaie and Hailesealsie Mekseb for their incomparable and indispensable encouragement, motivation, aspiration and advice in different aspects of my personal and academic life. My special thanks extend to my friends Arnoala, Kanta, Kassa, Salem and others.

I would like also to express my love and appreciation to my family for their unyielding love, support and prayers especially Kemma, Mussie, Sara, Berhane, Aster, Seyum, Emilia and Mehir.

At last, I would like to praise and grace almighty God in the name of Jesus Christ for giving me endless strength, love and energy in every step of my life.

Dedication

I dedicate this paper to my beloved mother Keima Ghebremeskel and late brothers Daniel Goitom, Mengis Goitom and Tedros Goitom.

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1 Introduction

The definition of an ecosystem in modern times was introduced by the British plant ecologist Transley (1935) and includes not only the community of organisms in an environment but also the whole complex of the physical and chemical factors around them (Voigt et al., 2007). Until recently the environment was taken for granted as the satisfier of all mankind's needs, providing food, mineral and biological resources and a seemingly limitless facility for the disposal of waste materials (Shaw 2007).Most ecosystems can't be regarded as having definite borders but a minimum of three distinct components (Voigt et al., 2007). Today, it is increasingly evident that the capacity of natural systems to absorb the wastes and contaminants of modern human existence is quantitatively and qualitatively limited (Shaw, 2007). So that understanding the flows of materials and energy within an ecosystem can help in quantifying the fluxes of radioactive substances which lead to irradiation of humans and other species (Voigt et al., 2007). Radionuclide substances are produced in small quantities but are potentially so detrimental that their presence in the environment even at low levels is sufficient to cause concern. Some of which are natural components of the environment but many of which are present largely as a result of man's activities (Shaw, 2007).

Radioactive contamination of the human environment became a reality on 16 July 1945 when the first fission was tested near the town of Alamogordo in New Mexico, USA. However, large-scale production of anthropogenic radionuclides began when the first nuclear reactor started operation in Chicago on December 2. 1942 (Aarkrog. 1994).Prior to the release of large quantities of radioactive fission and activation products into the environment during the atmospheric weapons tests 1945-1963, there was limited scientific interest in the environmental radioactivity as the world inventory of artificial radionuclides was very small (Shaw, 2007). Atmospheric testing of nuclear weapons was carried out in two main series from 1952 to 1958 and 1961 to 1962. The USSR, USA and UK signed the partial atmospheric test ban treaty in 1963 and thereafter, only a limited number of atmospheric tests were carried out by France, China and India (MacKenzie, 2000).

About 1300 different radionuclides exist partly originating from natural sources, and partly anthropogenically produced. Natural radionuclides can be generated by activation of stable isotopes via cosmic radiation for example ^{3}H (12.3 years of half-life), ^{7}Be (53.3 days), ^{14}C

(5730 years), ^{35}S (87.5 days) known as Cosmogenic radionuclides, which are originated during the creation of the universe. The latter are known as primordial radionuclides and include ^{40}K ($1.3*10^{9}$ years) and the isotopes of uranium and thorium which give rise to various daughter nuclides including ^{226}Ra (1600 years), ^{222} Rn (3.8 days), ^{210}Pb (22.3 years) (Strebl, 2007). Artificial radionuclides have entered the human environment as a consequence of atmospheric nuclear weapon tests and also by nuclear accidents. Global fallout of fission products in the 1950s and 1960s involved deposition of ^{137}Cs (30.17 years), ^{90}Sr (28.5 years), ^{89}Sr (50.5 days), ^{3}H, ^{54}Mn (312 days), ^{65}Zn (244 days), ^{95}Zr (64 days), $^{103/106}$ Ru (39.47368 days), ^{129} I ($1.6*10^{7}$ years) and ^{144}Ce (284.8 days). At present the most important sources are routine releases from nuclear power plants (NPP) and reprocessing plants. Very high emissions of these isotopes can occur in the very rare circumstances of nuclear power plant accidents such as occurred at Chernobyl (Ukraine) in 1986. This accident caused deposition of substantial quantities of radionuclides over a large part of Europe, and in Sweden four counties were mainly affected (Rosén, 1995). Hence, it increased the $^{134/137}Cs$ and ^{90}Sr soil inventories of Sweden and other many European countries considerably (Strebl et al., 2007). In USA, many nuclear plant accidents were occurred and caused fatalities and property damages. One of them is the accident happened in Idaho Falls, Idaho, January 3, 1961 where the damage costs around 22 million dollar.

In Japan, the Fukushima I nuclear accidents occurred after a 9.0 magnitude Tōhoku earthquake and subsequent tsunami on 11 March 2011, only 14 days before the reactor was to be shut down. The earthquake triggered a scram shut down of the three active reactors at the Fukushima I Nuclear Power Plant (Fukushima Dai-Ichi). The ensuing tsunami inundated the site, stopped the Fukushima I backup diesel generators, and caused a station blackout. The subsequent lack of cooling led to explosions and meltdowns at the Fukushima I facility, with problems at three of the six reactors and in one of the six spent fuel pools. It caused fatalities and properties damage and evacuation of millions of people.

2.0 Background

The natural and artificial radionuclides that are present in the environment are the main sources of radiation exposure for human beings and constitute the background radiation level (Bozkurt et al., 2007). Contamination of the environment has been a frequent legacy of

industrialization, and recognition of the adverse health, environmental and economic effects of this contamination resulted in legislation for its minimization and monitoring (Willey et al., 2007). Thus, determining the distribution of these radionuclides is necessary for assessing the effects of radiation exposure (Bozkurt et al., 2007). There is now increasing pressure to develop effective technologies not just to minimize and monitor but also to decontaminate ecosystem compartments such as soils that have become contaminated; the nuclear industry is similar to other industries in these respects (Willey et al., 2007).

Certainly, radionuclides in an environment will persist with more or less predictable residence times, and with associated radiation exposures in the proportion to the type and activities of radionuclides present (Shaw, 2007). Under natural conditions, all living organisms are exposed to background-radiation. The normal dose rate in the Nordic countries is about 1 mSv per year (Johansson. 1994). The sources of this radiation are decaying radionuclides in the ground and Radon (about 0.5 mSv per year), cosmic origin (about 0.3 mSv), and radiation from internal radionuclides (mainly ^{40}K. about 0.2 mSv).. In Sweden an additional mean annual dose of about 3 mSv is obtained from inhaled radon daughters (Johansson. 1994).

Radioactivity from the Chernobyl accident affected food production systems throughout Europe. Most affected were the Scandinavian countries where activity concentrations in reindeer, goat's milk, sheep, game animals and fresh water fish were above the intervention levels, are still subject to restrictions (Smith et al., 2007). Radio-cesium and radio-strontium were still present in the global environment at relatively low concentrations prior to the Chernobyl accident as the result of atmospheric nuclear weapons and discharges from natural facilities to a lesser extent (Smith et at., 2007). The global fallout of ^{137}Cs resulting from the atmospheric nuclear weapons testing during the 1960s has been estimated at 2.8 k Bq/m^2 in the Northern Hemisphere and 2.2 k Bq/m^2 in central Sweden (Rosèn et al., 1999). However from Chernobyl accident, they were 1.8 and 1.4 kBq/m^2 respectively (Rosèn et al., 1999).

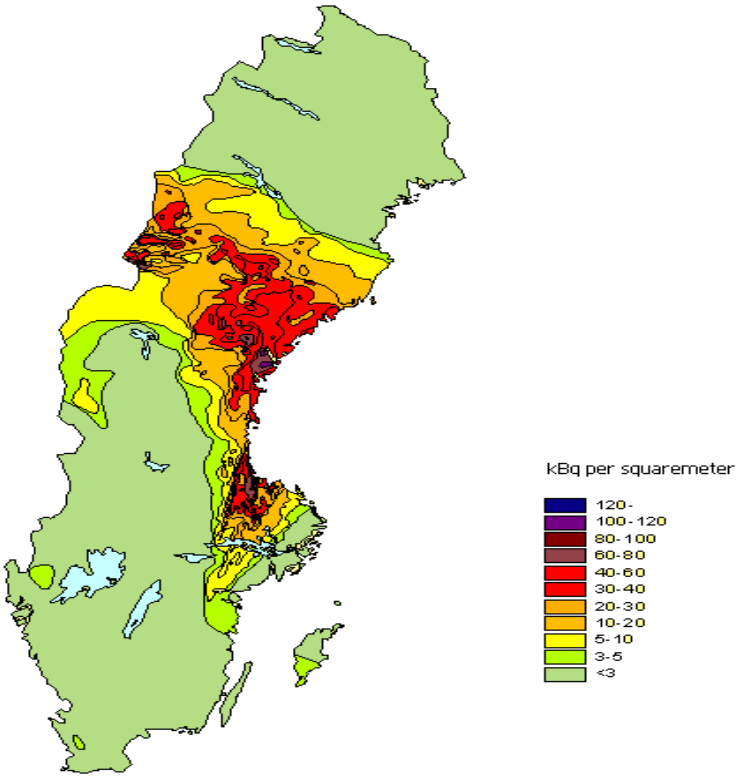

Figure 1 The deposition of 137 Cs in Sweden after the Chernobyl accident, and measurements were taken in May to October 1986 (Sveriges National atlas homepage, 2007 and Larsson 2008)

The total amount of ^{137} Cs deposited over Sweden has been estimated to be $4.25*10^{15}$ Bq (Åhman, 1994). The radio-cesium was mainly deposited in the middle and northern parts of the country (Ågren 1999). In some places, the Chernobyl fallout of ^{137} Cs in the Sweden was up to 100 times higher than the global bomb fallout (Rosèn, 1996). The greater activity of ^{137} Cs in the initial fallout, twice the activity of ^{134} Cs and its longer physical half-life, makes ^{137} Cs the more important of the two cesium isotopes (Åhman 1994 and Rosen 1996).

Plants are the primary recipients of radioactive contamination to the food chain from the abiotic environment through the uptake of radioactive debris from the atmosphere by above-ground parts of plants and a sorption of debris from the soil by the root system of plants (Aarkrog, 1994). Experience since the Chernobyl accident indicates that the intake of radioactivity through food is an important source of the dose to the population in the Nordic countries such as Sweden. In the long term, the problems will be connected to the contamination of various sensitive ecosystems (Strand, 1994). Plant development stage and Seasonality are the varying response to radioactive contamination of vegetation when the

contamination occurs. Short-lived radionuclides (as ^{131}I) and those that enter the food chain by direct contamination (e.g. ^{137}Cs) are important in this connection (Aarkorg, 1994).

2.1 General statement of the problem

Radionuclides from natural decay chains despite short physical half-lives can be of high radiological importance because they are continuously produced and thus remain in the environment at a constant level (Streble et al., 2007). Besides to the total amount deposited, the radiological importance of artificial radionuclides is determined by their radioactivity and radiation type (alpha, beta or gamma), their bioavailability and behavior within the food chain (Streble et al., 2007). Some radioactive isotopes like those of iodine cannot be discriminated from the stable forms and readily enter living systems. Others show close chemical similarity to nutrients (Cs-K; Sr, Ra-Ca), which can lead to a considerable uptake into the food-chain (Streble et al., 2007).

Irradiation of humans can occur via external and internal exposure to radionuclides (Voigt et al., 2007). Doses to humans are estimated by considering ingestion of radionuclides in drinking water and food, external irradiation from radionuclides in soil, and inhalation of radionuclides on airborne dust particles (Brennwald et al., 2009), and these account for a substantial part of the average radiation doses received by various organs of the human body (Khan et al., 2010 and Voigt et al., 2007).

Soil-plant-human is recognized as one of the major pathways for the transfer of radionuclides to human beings (Jabbar et al., 2010). Knowledge, description and modeling of radionuclide transfer in food chains are one of the key topics in radiation protection and radioecology (Voigt et al., 2007). Furthermore, the migration and accumulation of radionuclides in the soil plant system is a complex phenomenon, involving processes such as leaching, capillary rise, runoff, sorption, root uptake, and re-suspension into the atmosphere (Jabbar et al., 2010). Although the majority of the research to date has focused on the contamination of food products and its prediction, an attempt is made here to draw together the more ecological aspects such as the distribution and fluxes or radionuclides in the different compartments (Salt, 2007). In response, countermeasures to reduce especially radio-cesium, radio-iodine and radio-strontium in animal products have been studied, tested and implemented, including those for animals living in semi-natural environments. However, to apply countermeasures

most effectively, the behavior and transfer of radionuclides in animals need to be understood and properly modeled (Voigt et al., 2007).

Post-Chernobyl studies confirmed previous work showing the influence of soil properties (clay mineral content and exchangeable potassium concentration) on radio-cesium uptake by the food chain (Smith et al., 2005). The migration and distribution of radio-cesium in the soil profile varies depending on soil properties such as soil texture, organic matter content and pH, as well as on climatic conditions, land use and management practices. Important factors affecting the transfer of radio-cesium to crops/plants are the distribution of the root system in the soil profile as well as the soil pH and nutrient status (Rosén et al., 1999).

In Sweden, according to Rosen et al 1996, the results showed that agriculture (pasturing and fodder) is sensitive to transfer of radio-cesium where there was a need of counter measures for some farms to reduce the transfer to milk and meat. Such conditions may exist for a long time for the reason that the main part of 137Cs deposited on the ground will remain within the root zone for a long time (Rosén et al, 1996), and is readily available for root uptake and transfer to plants/crops (Rosén et al., 1999). The knowledge of the dynamics of radio-cesium and natural radionuclides deposition on the soil surface and its downward migration through the soil is important for predicting the external radiation dose and internal radiation through ingestion (Rosén et al., 1999; Khan et al., 2010).

3.0 Aim of the study

The study has three main objectives:

1. To assess the migration of natural and artificial radionuclides (^{137}Cs) in soil profile of Uppsala and Jämtland study areas
2. To measure the activity concentrations and transfer factors of radionuclides from soil-to-plant and from plant to milk
3. To carry out dose assessment of the radioactive substances by comparing the in situ and laboratory measurement of radioactivity concentration of radionuclides in the study areas.

4 General concepts

4.1 Radioactive substances

Primordial radionuclides (^{238}U, ^{235}U, ^{232}Th and ^{40}K) were formed by the same stellar processes which formed the other heavy elements of the Earth (Smith and Nicholas, 2005). They are long-lived species which have been present on the earth since its formation 4.5 10^9 years ago (MacKenzie, 2000 and Choppin et al., 2002). The radionuclides ^{238}U, ^{232}Th and their decay products, and ^{40}K are natural terrestrial radionuclides and significantly form of the major part of the natural radiation dose (Smith and Nicholas, 2005 and Karahan et al., 1999). Due to its low natural abundance, ^{235}U and its decay products do not form a significant part of natural radiation exposure (Smith and Nicholas, 2005). The primordial radionuclides ^{238}U, ^{235}U, and ^{232}Th are the parent members of the three natural radioactive decay series, and their main members along their half-lives and principal decay modes are shown below (MacKenzie. 2000):

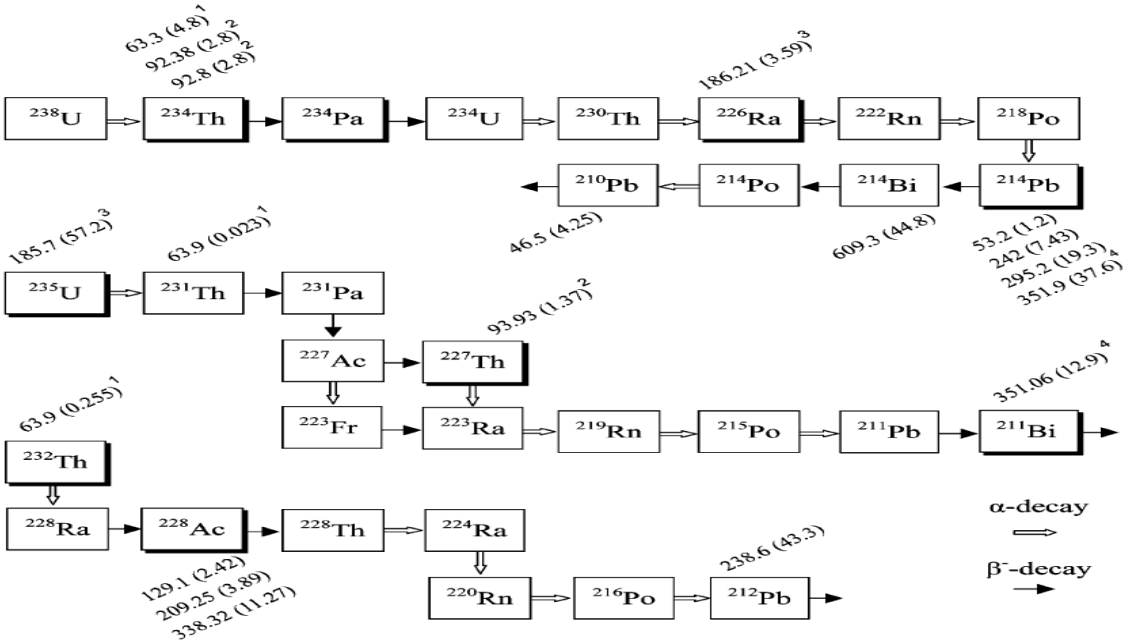

Figure 2 ^{238}U, ^{232}U and ^{232}Th series.

Radiation doses from primordial radionuclides are primarily from external gamma radiation, ingesting and inhalation. Both external and internal doses can vary significantly according to differences in the geology of a region (Smith and Nicholas, 2005).

4.2 Ionization Radiation

Ionizing radiation is energy in the form of waves or particles that has enough force to remove electrons from atoms (EPA, 2007). One source of radiation is the nuclei of unstable atoms where the nuclides disintegrate in a process where one element is transformed into another, the excess energy of the nuclide being dissipated as electromagnetic radiation or particles. As a consequence, radiation is transported as energy (Thored, 2010 and EPA, 2007). The major types of radiation emitted during the radioactive decay (ionizing radiation) are alpha, beta and gamma radiations (Mobbs, 2009, Thored, 2010 and EPA, 2007).

4.2.1 Alpha radiation

Alpha radiation (particles) is energetic, positively charge particles consisting of two protons and two neutron (Helium nucleus) (EPA, 2007 and Thored, 2010).Alpha radiation does not travel very far (Mobbs 2009) and its spreading range is a couple of centimeters (Mobbs. 2009 and Tored. 2010). The alpha particle is a stable unit and when it hits the surrounding media, it picks up two electrons and become an uncharged helium atom (Thored. 2010). Even though they are highly energetic, the high mass of alpha particles makes them to move slowly through the air (EPA, 2007). External exposure is of far less concern than internal exposure, because alpha particles lack energy to penetrate the outer dead layer of skin (EPA. 2007). Internal exposure through inhalation, ingestion, or absorption into blood stream could be very harmful where sensitive tissue is exposed to alpha radiation (USEPA. 2007). Examples of alpha radiation are ^{238} U, ^{226} Ra, and ^{210} Po. An example of α decay is as follows:

$$^{226}\text{Ra} \longrightarrow {}^{222}\text{Rn} + \alpha$$

4.2.2 Beta radiation

Beta particles are fast moving electrons emitted from the nucleus during the radioactive decay (USEPA. 2007). Example of Beta emitter radioactive elements are Iodine, Strontium-90, Carbon 14, Cesium etc., (Thored, 2010 and EPA, 2007). Beta radiation is constituted by an electron (Beta-) or a positron (Beta +). They are more penetrating than alpha particles (EPA. 2007 and Mobbs. 2009). Beta emitters like alpha emitters, are most hazardous when they are inhaled or ingested (EPA 2007). An example of beta radiation is ^{214} Pb decay to ^{214} Bi: ^{214} Pb $\longrightarrow {}^{214}$ Bi + β

4.2.3 Gamma Radiation

Gamma rays are packets of photons of energy (EPA, 2007). Photons are electromagnetic radiation emitted with longest range and they can travel hundreds of meters in air (Thored, 2010). They are often accompanying the emission of alpha or beta particles from a nucleus. They don't have charge and mass, and are very penetrating (EPA, 2007). The examples of gamma radiation radioactive elements are Cesium-137, Cobalt-60 etc. They are a radiation hazard to entire body, and can easily pass through the human body where a fraction will always be absorbed by tissue (Mobbs, 2009 and EPA, 2007). An example of gamma emitter is ^{60m} co which decays and emits gamma rays:

$$^{60m}\text{Co} \longrightarrow {}^{60}\text{Co} + \gamma$$

4.3 Half-life

There are three types of mechanisms of estimating half-life of radionuclides. These are Physical, Biological and Ecological half-life. In this study, it is used only the physical half-life. The physical half- life of radioactive element is defined as the time taken for one-half of a given amount of a radioactive element to decay (i.e. to transport in to another isotope or element by emission of radiation). The activity concentration of a radionuclide at time t is given by:

$$C(t) = C(0) \exp(-k_p t) \quad (1)$$

Where C (0) is the activity concentration at time t = 0 and k_p is the radioactive decay constant. The radioactive decay constant is related to the physical half-life T_p by:

$$T_p = \ln 2/k_p \quad (2)$$

4.4 Migration and Transfer factor

For radionuclides present in terrestrial ecosystem, information about their migration in soils is crucial, since this process controls their long-term behavior in the environment, their uptake by flora and fauna including food chains, but also their potential as groundwater contaminant (Kirchner, 2008). In soil, the type of radionuclide element determines only their potential ability to migrate through the profile (Sokolik et al., 2001). In soil profile, it is

believed that the initial movement of deposited radioactive elements or radionuclides from the soil surface is relatively rapid as the result of infiltration process (Smith and Nicholas, 2005).And, the radionuclide migration intensity is observed as the result of simultaneously occurring processes: transport caused by rainfall infiltration (convective transport), dispersion caused by spatial variations of convective velocities, transfer on colloidal and fine-dispersed particles (diffusion process), migration along the plant root system and others (Sokolik et al., 2001 and Kirchner et al., 2008). In addition to these abiotic processes, soil fauna may contribute to the transport of radionuclides in soils but their action under general conditions results in a dispersion- like translocation (Kirchner et al., 2008).Hence, the vertical migration of the radionuclides is much slower as a significant amount of the fallout of the majority is sorbed to the soil matrix (Smith and Nicholas, 2005). However, for some radionuclides such as radio-cesium, their mobility might be decreasing over period of time after fallout due to the presence of slow sorption reactions in soil (Smith and Nicholas, 2005). Thus, the physical and chemical behaviors of the fallout (radionuclides) influence its migration in the soil (Smith and Nicholas, 2005). The vertical migration of radionuclides in soils has a direct effect to the levels external radiation exposure to the people living or working in areas contaminated by radioactive elements. The concentration (amount) and distribution of radioactive elements in the root zone of plants also affects to the transfer of radioactivity to the food chain and the transport of radioactive elements from contaminated soil to plants (Smith and Nicholas, 2005).The factors that determine the variation of radionuclide activity with depth in soil profile are rainfall intensity, pH, organic matter content, soil moisture content, soil texture, soil structure, infiltration rate, sorption characteristics of the particular radionuclide, land use and management practices (Rosen et al., 1998 and Smith and Nicholas, 2005).

Due to the inherent complexity and spatial variability of the soil-pant system, the uptake of radionuclides in vegetation from soil is difficult to quantify. The Transfer Factor (TF) is a useful parameter, which is usually used for evaluating the impact of releases of radionuclides in to the environment (Jabbar et al., 2011). The TF depends on the vegetation type, soil properties and the type of radionuclides (Smith J. and Beresford N. A, 2005).

5 Methodology

5.1 Radioactive equilibrium

Radioactive equilibrium concept was applied to measure the activity of nuclides in soil, grass and milk powder. There are three kinds of equilibrium based on the time required to reach equilibrium by comparing both half-life of a parent and its daughter of radionuclide element. These are Secular equilibrium, Transient equilibrium and No equilibrium. In this study, a secular equilibrium was used to measure the activity of radionuclides in soil, grass and milk powder. In the case secular equilibrium, the half-life of the parent (1) is much longer than that of the daughter (2). Thus the evolution of the number of atoms of the daughter nuclei (N_2) is described by the following equation:

$$N_2 = (k_1/k_2)*N_1\ (0)*\ (1 - e^{-k2t}) \qquad (3)$$

Where N_1 is the number of atoms of parent nuclei, t is time, K1 is decay constant of parent and K_2 is the decay constant of daughter's nuclei.

5.2 Study area

The study areas are precisely described in table 1. And, the type of soil and their physical and chemical characteristics of the study areas are described in detail in Annex 1.

Table 1 The description of study areas in Uppsala and Jämtland Counties.

Place	**Hallen**	**Möjsjövik**	**Lövstalöt**	**Myrviken**	**Backfors**	**Vikdrolet**
Location	63º10'5"N. 14º 7' 13" E	59°56'60"N.17 °13' 60" E	59°57'10"N.17° 35' 0" E	62º59'11.0''N. 14º 24' 26" E	63º01'10.3''N. 14º 21' 50" E	63º00'55.5'' N.14º23'0.23" E
Mean temperature	2.5 ºC	4.7 ºC	5.3 ºC	2.5 ºC	2.5 ºC	2.5 ºC
Mean precipitation	549 mm/yr	583 mm/yr	535 mm/yr	549 mm/yr	549 mm/yr	549 mm/yr
Grazing period (start -end)	01/06/2010 – 01/09/2010	15/05/2010 – 15/09/2010	15/05/2010 – 15/09/2010	01/06 - 01/06	-	01/06 - 01/06
Number of animals grazing	120	10	140	100	-	<20
Type of breed	Swedish lowland livestock. (SLB). Cow	horses	Swedish lowland livestock. (SRB). Cow	Swedish lowland livestock. (SLB). Cow	-	-
Milk production (l/cow/year	9500	-	9000	-	-	-
Type of farm	intensive	extensive	intensive	intensive	extensive	extensive
Land Use	Ecological farming	Not ploughing since 1986	Ecological farming			

Table 1 gives an overview of the study areas by describing the location, land use, land availability, farming system, and agricultural production.

5.3 Sampling technique

At each location, the soil cores with a diameter of 21 mm were sampled to a depth of 25 cm and sliced in to each 5 cm layer, and taken one sample from each sub-site marked with a distance of 20 meter away from the center of the circle (see figure 3). The soil samples from the same soil layer of the sub-sites were mixed to form a representative bulk soil sample of each layer. Hence, replicate samples could be maintained for statistical analysis.

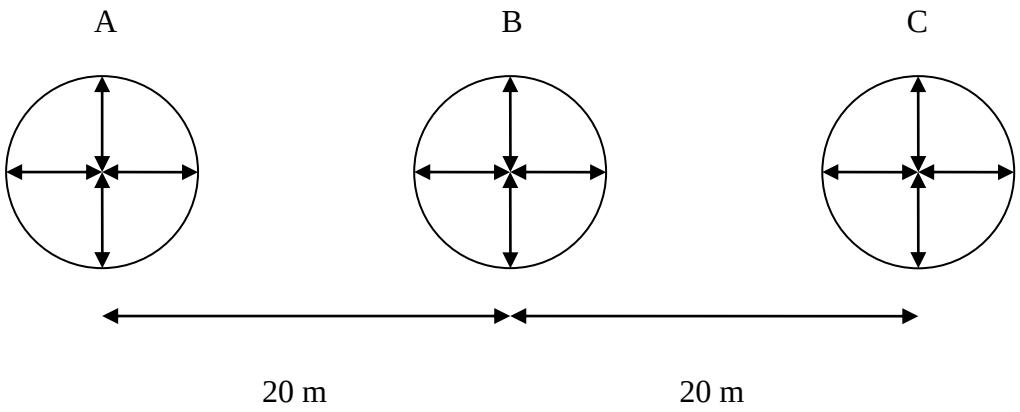

Figure 3 Schematic sketch of sampling technique: in each circle, 12 core samples with a diameter of 21 mm auger to a depth of 25 cm were collected and sliced to a depth of 5 cm layer within the circle a diameter of 6 m.

The bulk soils were dried out in air-dried room with a temperature of maximum 40 °C for one week. They were crushed and grounded to powder; mixed and passed through 2 mm sieve for removal of stones and roots. Then, representative soil samples were prepared in a petri dish with a known weight and covered to maintain a closed system and kept for 30 days to attain a secular equilibrium before radiometric analysis was carried out.

At the sampling location, grass samples were collected at each site four times within an area of 1 m^2 rectangle to form a replicate grass sample and taken at the same date as the soil samples. The grasses and herbs were cut around 5 cm above the ground. Grass samples were dried in a drying machine at 60 °C of temperature for 4 days. Then, they were milled to prepare a sample in a 60 ml petri dish with a known weight and covered to maintain a closed system for 30 days to attain secular equilibrium before radiometric analysis was carried out.

5.4 Sample treatment of milk

At each location, milk samples were directly taken from the cow. They were kept in a defreeze room (-18 to -20 °C) to avoid any contamination. In order to have a better detection limit, the liquid milk samples were converted in to powder after crushing them into pieces and put them in a freeze-drier machine for more than one week time. Appropriate cautions were taken to avoid any contamination between the samples by cleaning the machine properly at each sample. Similar to soil samples, milk powder samples were prepared in 60 g

plastic petri dish and covered to maintain a closed system and kept in a cooling room for 30 days to attain a secular equilibrium before radionuclides were detected and analyzed.

5.5 Measurement Technique

All samples were analyzed by using two high purity germanium detectors (HPGe).A petri dish containing 60 g of soils, grass and milk powder, was used for measuring activities of ^{40}K, ^{226} Ra, ^{232} Th, ^{210} Pb, ^{238} U and ^{137} Cs. The sample containers were sealed with parafilm to maintain a closed system (avoid any leakage/escape of radionuclides) and left minimum for 30 days in order to attain a secular equilibrium between the above mentioned radionuclides (except ^{40}K) and their gamma-emitting daughters such as ^{234}Th (daughter of ^{238} U), ^{228} Ac (daughter of ^{232}Th), ^{214} Bi (daughter of ^{222}Rn), and ^{222}Rn (daughter of ^{226}Ra). The minimum measurement period for grass samples were 3 days (72 hours) but for soil and milk powder samples were only one day (24 hours). The activity uncertainty and minimum detection activity (MDA) of those radionuclides were also measured. The detector was calibrated every 3-4 weeks' time by running it for several days Energy calibration was made using a certified radioactive element with known standard activity. Figure 4 shows the result of the energy calibration performed in one of the detectors used in the work.

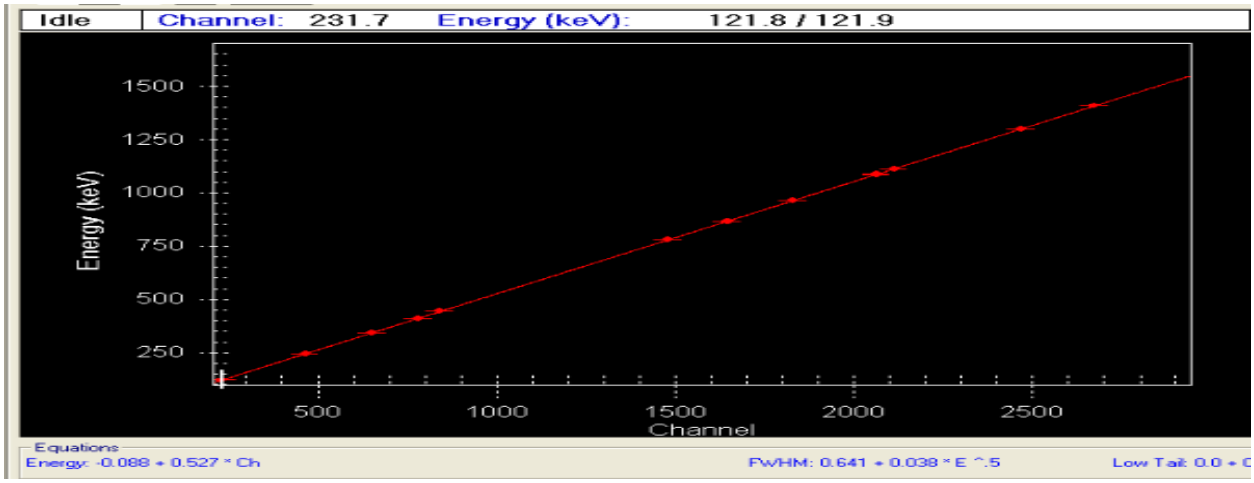

Figure 4. An example of energy calibration of a detector used in this study.

The Efficiency of the detector was calibrated using the same geometry of the container and the same density for each type of the sample used in the study (Figure 5).

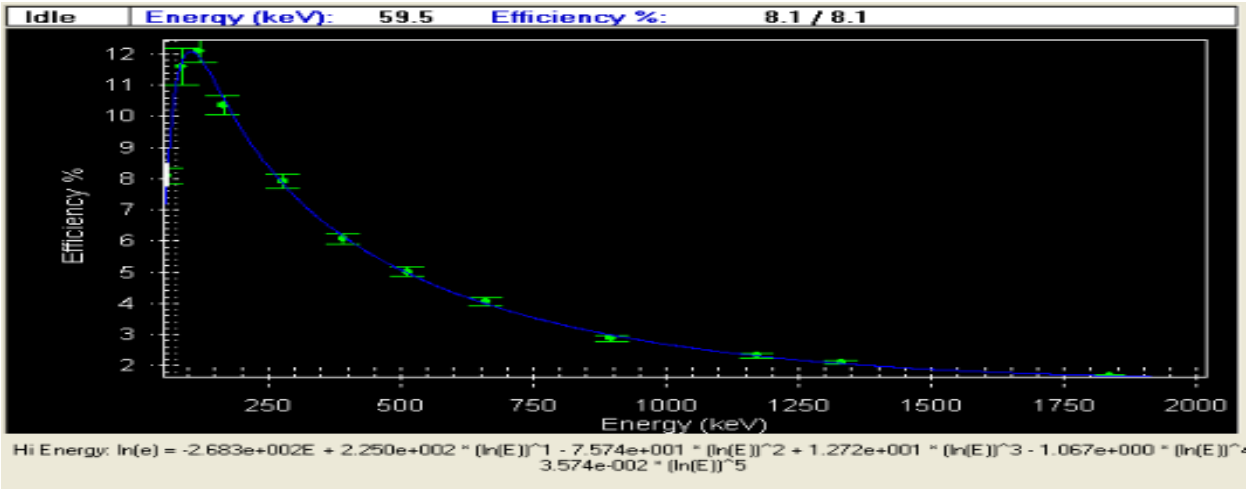

Figure 5. An example of efficiency calibration of a detector used in this study.

5.6 Transfer factor

i. Transfer factor of artificial radionuclides (^{137}Cs) from soil to grass (TFg), and from grass to milk

1. $$TFg\left(\frac{m2}{kg}\right) = \frac{\text{Activity concentration of 137 Cs in grass dry matter } (\frac{\text{Bq}}{\text{kg}}))}{Activity\ concentration\ of\ \ 137\ Cs\ depostion\ in\ soil\ per\ unit\ area\ \left(\frac{Bq}{m2}\right)} \quad (4)$$

TF Grass is m^2/kg

1. $$TFm(\frac{m2}{kg}) = \frac{\text{Activity concentration of 137 Cs in milk powder } (\frac{\text{Bq}}{\text{kg}}))}{Activity\ concentration\ of\ \ 137\ Cs\ depostion\ in\ grass\ (\frac{Bq}{m2})}$$

(5)

TF Grass is dimensionless.

ii. Transfer factor of natural radionuclides from soil to grass (TFg), and from grass to milk

1. $TFg =$

$$\frac{\text{Activity concentration of radionuclides in grass } (\frac{\text{Bq}}{\text{kg}}))}{Activity\ concentration\ of\ radionulcides\ in\ soil\ (\frac{Bq}{kg})} \quad (6)$$

TF Grass is dimensionless.

2. $TFm =$

$$\frac{\text{Activity concentration of radionuclides in milk powder } (\frac{\text{Bq}}{\text{kg}}))}{Activity\ concentration\ of\ radionulcides\ in\ grass\ (\frac{Bq}{kg})} \quad (7)$$

TF Milk is dimensionless.

5.7 Dose assessment

5.7.1 Absorbed dose rate

The outdoor external absorbed dose rate (AD) is at 1 meter above the ground level can be calculated from activities of terrestrial radionuclides according to the following equation (Kumar et al., 2007):

$$AD = 0.462\ C_{Ra} + 0.608\ C_{Th} + 0.042\ C_K \quad (8)$$

C_{Ra}, C_{Th}, and C_K are the concentrations of ^{226}Ra, ^{232}Th, and ^{40}K in Bq/kg respectively.

5.7.2 Annual effective dose rate (AED)

The annual effective dose rate to the population can be estimated using the conversion coefficient from the absorbed dose in the air to effective dose (0.7SvG y^{-1}), taking into account the outdoor occupancy factor (0.2), and the indoor occupancy factor (0.8) (Huy et al., 2005). The study concerns only for the average annual effective dose of outdoors and could be calculated as follows:

Outdoors AED (mSv y^{-1}) = AD (nGyh-1) * 8760 h * 0.7 SvGy-1 * 0.2 (9)

The conversion factor of ^{137}Cs (Bq/m^2) into annual effective dose rate (μSv/y) is 1.40E-07 (Al, Hamarneh et al., 2002). The total annual effective dose rate (AED) is the sum of the annual effective dose rate of natural radionuclides and the annual effective dose rate of ^{137}Cs (μSv/y).

6. Results

6.1 Uppsala County

6.1.1 Activity concentration in soil and grass

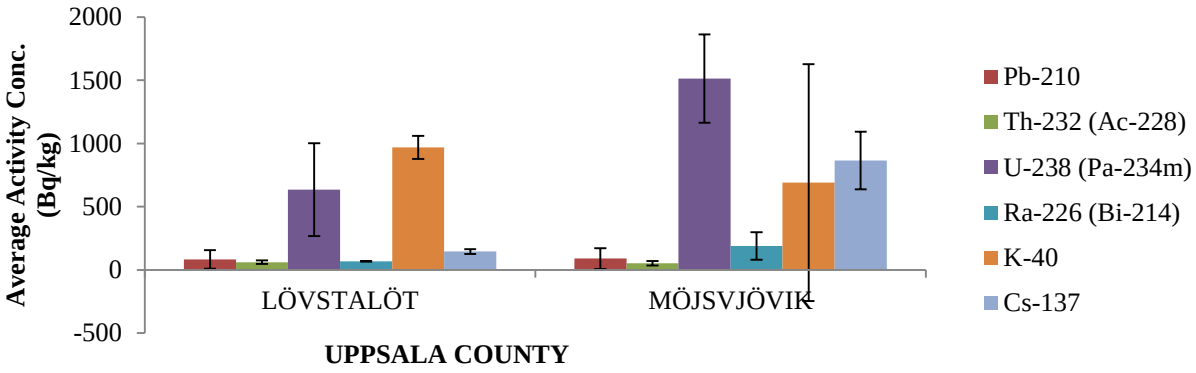

Figure 6 The average activity concentration of radionuclide elements at 5 cm depth of soil profile in Uppsala County

Figure 6 illustrates that the highest average activity concentration of radionuclide elements in soil in Lövstalöt study area was ^{40}K followed by ^{238}U and ^{137}Cs. Whereas in Möjsjövik was ^{238}U followed by ^{137}Cs and ^{40}K, subsequently. The average activity concentration of ^{238}U, ^{137}Cs and ^{226}Ra in Möjsjövik are higher than in Lövstalöt. But ^{40}K was higher in Lövstalöt than in Möjsjövik.

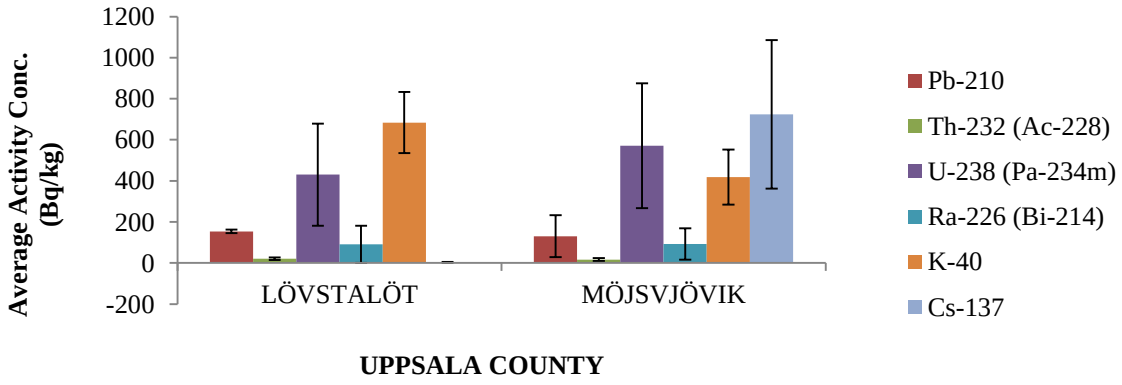

Figure 7 The average activity concentration of radionuclide elements in grass in Uppsala County

Figure 7 illustrates that the highest average activity concentration of radionuclide elements in grass in Lövstalöt study area was followed by ^{238}U and ^{210}Pb. In Möjsjövik, ^{137}Cs was the

highest followed by ^{238}U and ^{40}K. The average concentration of ^{137}Cs and ^{238}U were higher in Möjsjövik than in Lövstalöt but ^{40}K is higher in Lövstalöt than in Möjsjövik.

6.1.2 Activity concentration in milk

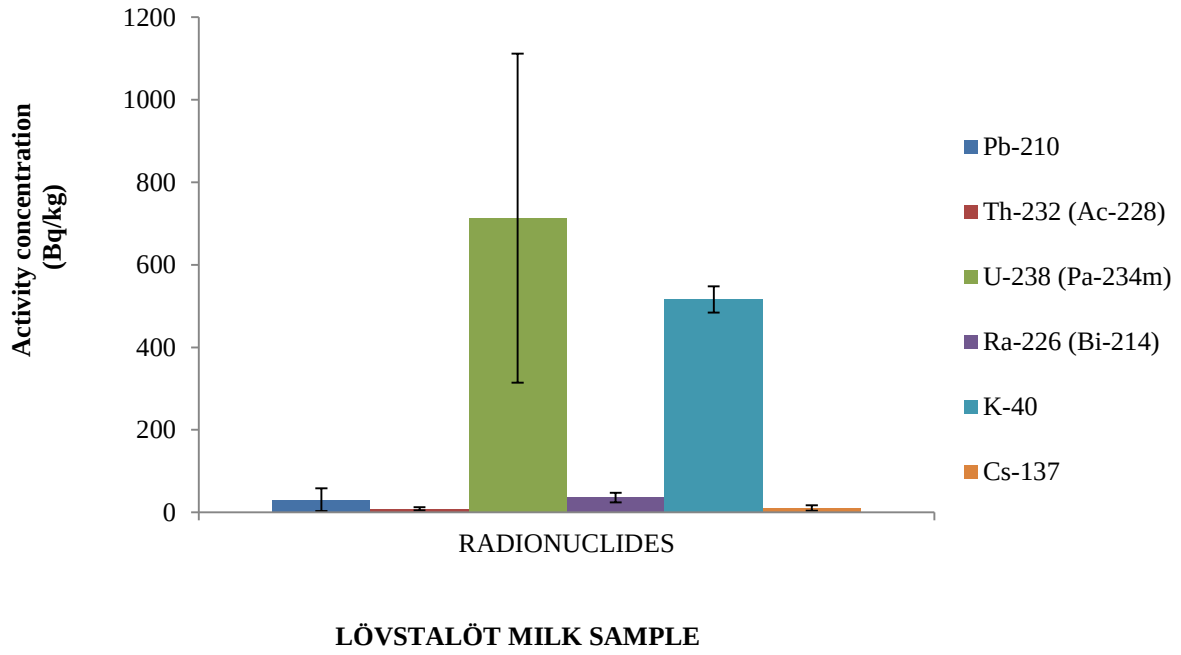

Figure 8 The activity concentration of radionuclide elements in milk in Lövstalöt, Uppsala County.

Figure 8 shows the activity concentration of radionuclide elements in milk in Lövstalöt study area. And, it was estimated that ^{238}U is the highest (around 750 Bq/kg) followed by ^{40}K (around 460 Bq/kg). However, the rest were very small comparing with ^{238}U and ^{40}K.

6.1.3 Transfer factor

Table 2: The average transfer factor of radionuclide elements in Uppsala County

Radionuclide elements		Study area	
Average Transfer Factor		**Lövstalöt**	**Möjsjövik**
K-40	Soil to grass	0.71 ± 0.193	2.72 ± 3.37
	Grass to milk	0.75±0.05	-
	Fodder to milk	-	-
Cs-137	Soil to grass	0.01±0	1.25±1.044
	Grass to milk	3.6±2.12	-
	Fodder to milk	-	-
Pb-210	Soil to grass	1.32 ± 0.8	0
	Grass to milk	0.14±0.23	-
	Fodder to milk	-	-
Ra 226 (Bi)	Soil to grass	1.36 ± 1.37	0.632± 0.42
	Grass to milk	0.31±0.27	-
	Fodder to milk	-	-
U-238 (pa-234)	Soil to grass	0.7±0.40	0.312± 0.16
	Grass to milk	0.87±1.50	-
	Fodder to milk	-	-
Th-232 (Ac-228)	Soil to grass	0.37 ± 0.19	0.407±0.18
	Grass to milk	0.00	-
	Fodder to milk	-	-

Table 2 shows the average transfer factor of radionuclide elements from soil to grass, grass to milk and fodder to milk in Uppsala County.

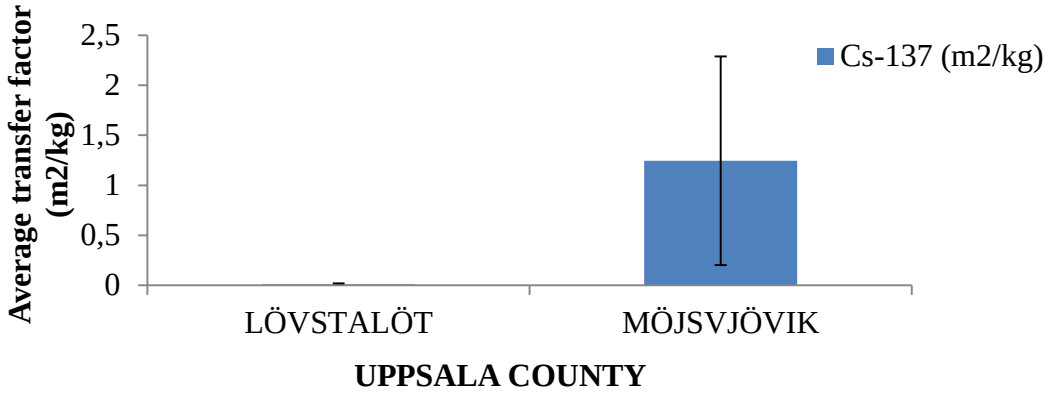

Figure 9 The average transfer factor of Cs-137 from soil to grass in Uppsala County.

From figure 9, it can be seen that the transfer factor of ^{137}Cs from soil to grass was significantly higher in Möjsjövik study area than in Lövstalöt. In Lövstalöt, the transfer factor was almost zero whereas in Möjsjövik is approximately 1.25 m^2/kg with high uncertainty.

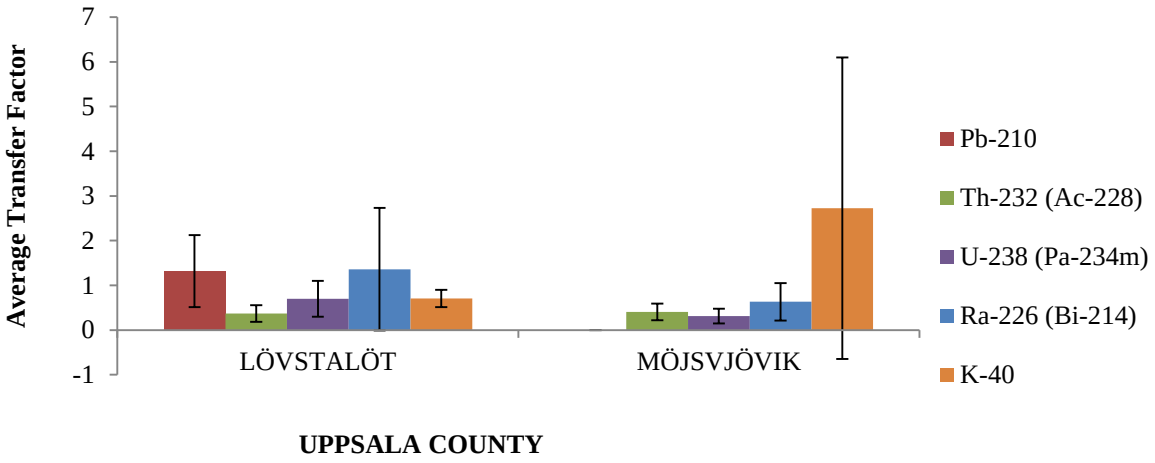

Figure 10 The average transfer factor of natural radionuclides from soil to grass in Uppsala County.

Figure 10 shows the average transfer factor of natural radionuclides from soil to grass in Lövstalöt and Möjsjövik study areas. In Lövstalöt, ^{210}Pb and ^{226}Ra have the highest transfer factor while in Möjsjövik was ^{40}K (around 2.4) with high significance and uncertainty followed by ^{226}Ra. When comparing the two study sites, the transfer factor of ^{40}K was higher in Möjsjövik but ^{210}Pb and ^{226}Ra were lower than in Lövstalöt.

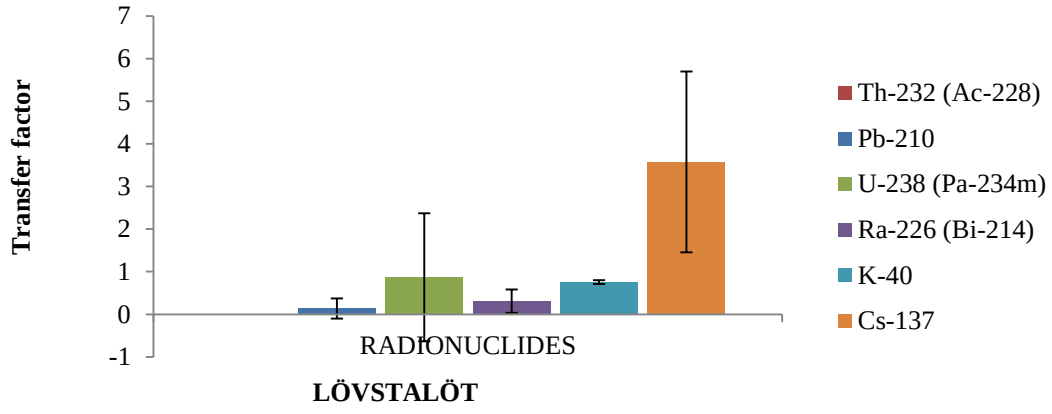

Figure 11 The transfer factor of radionuclides from grass to milk in Lövstalöt study area, Uppsala County.

Figure 11 illustrates that ^{137}Cs had significantly the highest estimated transfer factor of all the concerned radionuclide elements from grass to milk in Lövstalöt area (3.2). And, ^{238}U and ^{40}K had become the second and the third highest estimated transfer factor, respectively.

6.1.4 Soil profiles

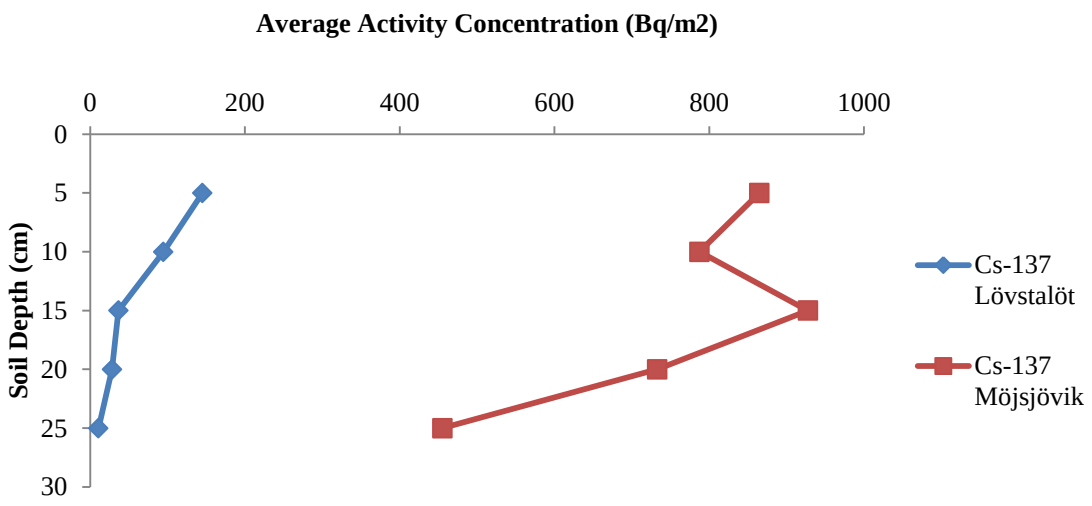

Figure 12 The migration of ^{137}Cs in the soil profile within the depth of 25 cm in Lövstalöt and Möjsjövik study areas.

Figure 12 shows the average activity concentration of ^{137}Cs in Lövstalöt. The average concentration of ^{137}Cs in soil profile of Möjsjövik was decreasing throughout the profile except at 15 cm depth. Thus, the trend fluctuation in the soil profile was attributed to the vertical downward and upward movement of ^{137}Cs; even lateral movement could be probably occurred. However, in the case of Lövstalöt, the average activity concentration of ^{137}Cs was decreasing with depth in the soil profile. A sharp decrease in the concentration was observed up to 15 cm depth followed by slight decrease at the rest depth of the profile. In general, the migration of ^{137}Cs in Lövstalöt is vertical downward in the profile.

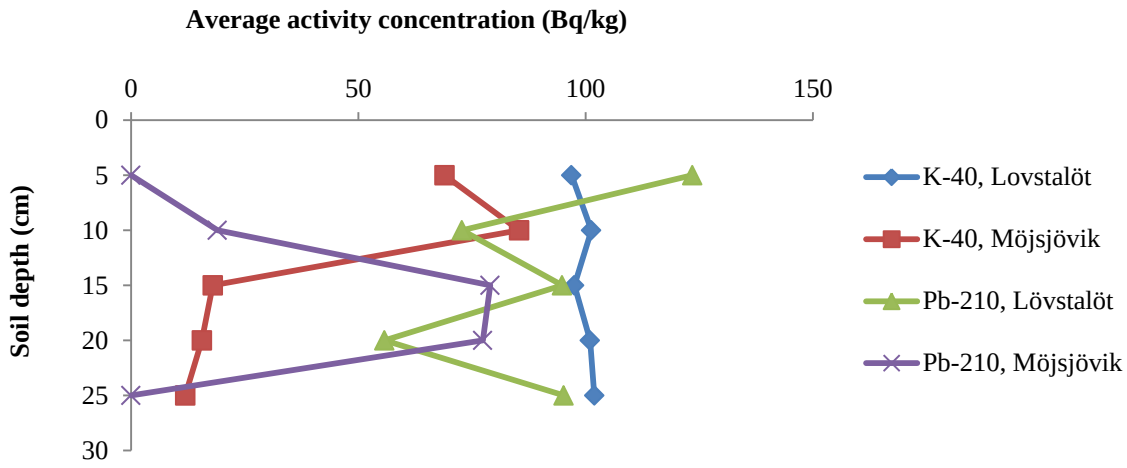

Figure 13 The distribution of ^{40}K and ^{210}Pb in the soil profile within the depth of 25 cm in Lövstalöt and Möjsjövik study areas.

In figure 13, the values of the activity concentration of ^{40}K are in a scale 10 times lower than the actual values in the profile. This was made to compare the trends of the radionuclides in the study areas. In Lövstalöt, ^{40}K was almost homogenous and uniformly distributed in the soil profile but in Möjsjövik, it fluctuates in the soil profile which indicates that it is not uniformly distributed and not homogenous in the profile. However, ^{210}Pb was not uniformly (non-homogenous) distributed in the soil profile.

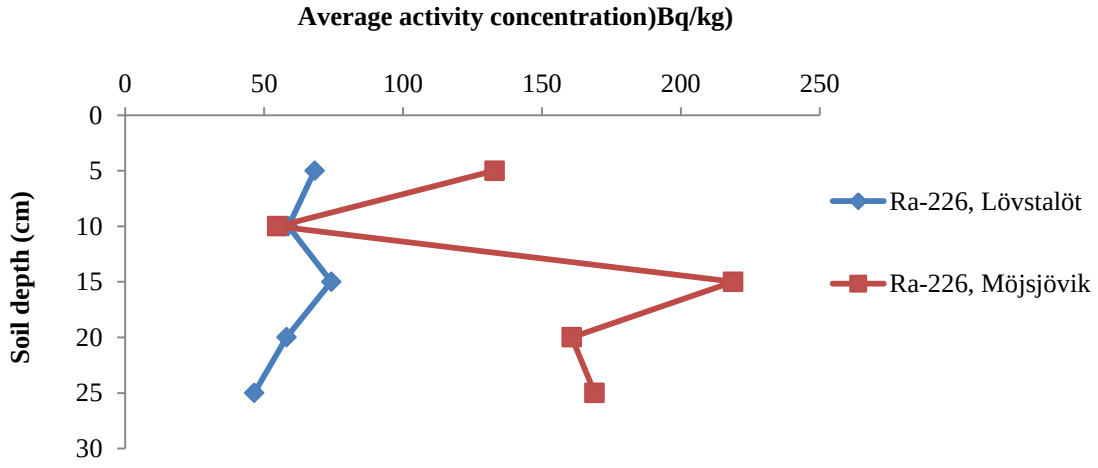

Figure 14 The distribution ^{226}Ra in the soil profile with a depth of 25 cm in Lövstalöt and Möjsjövik study areas.

Figure 14 demonstrates that the average activity concentration of ^{226}Ra in Möjsjövik was fluctuating very much in the soil profile, which indicates that there was a non-uniform (not

homogenous) distribution in the profile. Whereas in Lövstalöt the distribution was poorly uniform (homogenous) in the profile.

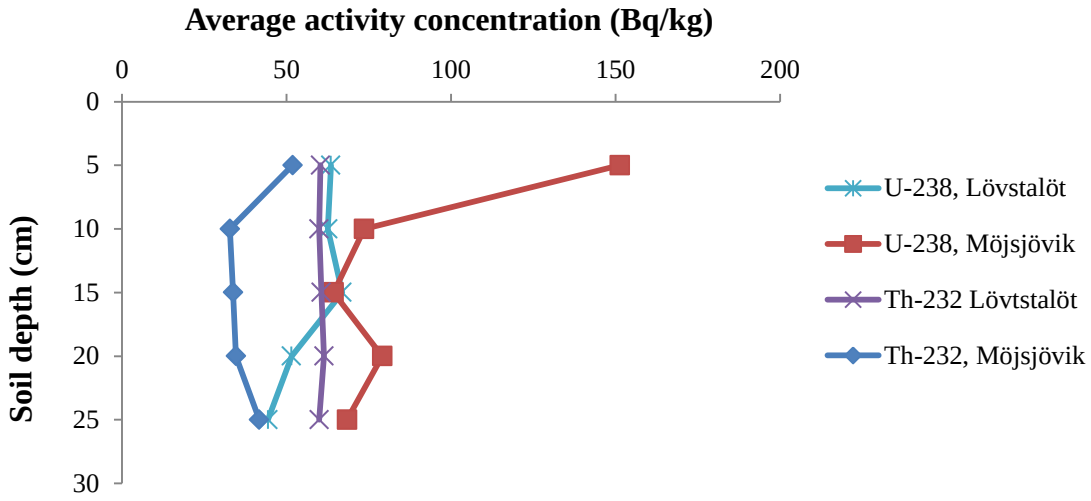

Figure 15 The migration of ^{238}U and ^{232}Th in the soil profile with a depth of 25 cm in Lövstalöt and Möjsjövik study areas.

In figure 15, the values of the average activity concentration of ^{238}U was divided by a factor 10 in order to obtain fitful graph with good trend and easier to make comparison between study areas. ^{232}Th in Möjsjövik was uniformly distributed in the soil profile. However, in Möjsjövik, the distribution of ^{238}U was non- uniform with high fluctuation in the soil profile. It is also non uniform in Lövstalöt, but less fluctuation in the profile comparing with Möjsjövik.

6.1.5 Statistical analysis- Regression analysis and Correlation coefficient

The regression analysis results, P-value (0 in Lövstalöt and 0.01 in Möjsjövik) and R-square (94.8% in Lövstalöt and 93.5% in Möjsjövik) show that both are statistically very significant with Lövstalöt being a bit stronger. These demonstrate that average activity of concentration of radionuclides in grass was strongly dependent on the average activity concentration of radionuclides in soil. In addition, both study areas had shown that the average activity concentration of radionuclides in grass was positively correlated with that in soil. The correlation was much stronger in Lövstalöt (0.973) than in Möjsjövik (0.874).

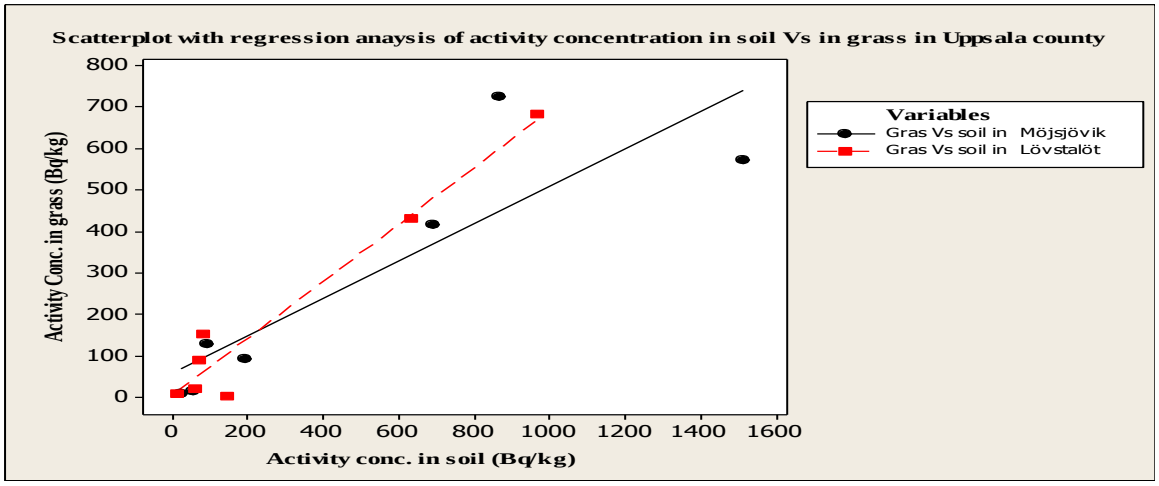

Figure 16 Scatterplot with regression analysis of the average activity concentration in soil versus in grass in Uppsala County.

Figure 16 illustrates the regression analysis of average activity concentration of radionuclides in grass versus in soil in Uppsala County. It displays that Lövstalöt study area had a better fitted line with closer residuals to the linear line than Möjsjövik.

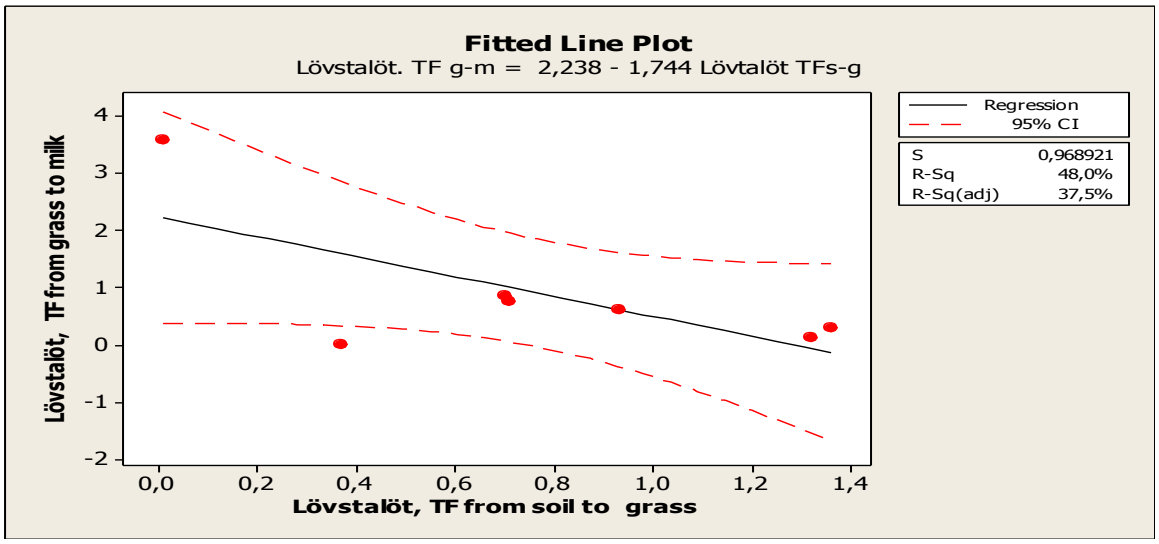

Figure 17 Fitted line plot with regression analysis of the TF from soil to grass versus the TF from grass to milk in Lövstalöt study area, Uppsala County.

Figure 17 shows the regression analysis results. With 95 % confidence interval, the obtained R-square and P-value were 0.48 and 0.085, respectively. The results explain that there is no statistical evidence to show that the TF from grass to milk was dependent on the TF from soil to grass.

6.1.6 Dose assessment

Table 3: The annual effective dose rate of natural radionuclides and ^{137}Cs (in situ and laboratory measurement).

Type of Measurement	Study Area	Absorbed dose rate of natural (nGy/h)	Annual effective natural radionuclide (µSv/y)	Annual Effective Dose rate of Cs-137 (µSv/y)	Total Annual Effective Dose Rate (µSv/y)
Lab.	Lövstalöt	1.1E+02	1.3E+02	3.8E+01	1.7E+02
	Möjsjövik	1.2E+02	1.5E+02	4.8E+02	6.3E+02
In situ	Lövstalöt	7.9E+01	9.7E+01	4.3E+01	1.4E+02
	Möjsjövik	1.8E+01	2.3E+01	2.4E+02	2.6E+02

Table 3 describes the annual effective dose rate of natural radionuclides and ^{137}Cs in situ and labaratory measurements in Lövstalöt and Möjsjövik study areas. In the case of natural radionucldies , first the annual absorbed dose rate was calculated using equation (8) and then the annual effective dose rate (equation 9). In both labaratory and in situ measurements, Möjsjövik (6.3E+02 and 2.6E+02 µSv/y) has higher total annual effective dose rate than Lövstalöt (1.7E+02 and 1.4E+02 µSv/y). In addition, the total annual effective dose rate estimated in the labaratory is higher than that of in situ in both study areas.However, the estimated annual absorbed dose rate of natural radionuclides in situ is higher in Lövstalöt than in Möjsjövik study area.

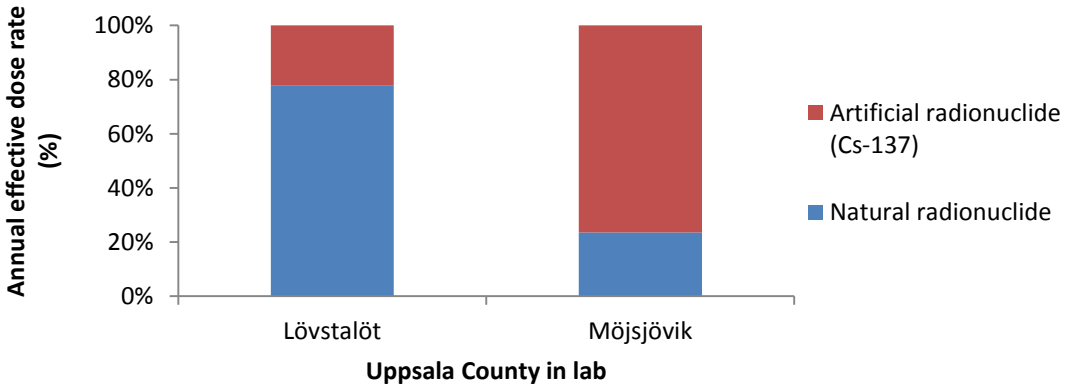

Figure 18 A comparison of percentage contribution of artificial and natural radionuclides to the annual effective dose rates in laboratory measurement of Uppsala County.

Figure 18 illustrates the laboratory analysis of the annual effective dose rate of natural and artificial radionuclides (in outdoor air at height of 1 m from ground surface) in Lövstalöt and Möjsjövik study areas. In Lövstalöt study area, the contribution of natural radionuclides to annual dose rate is almost 80% which is significantly higher than the artificial one (^{137}Cs), 20%. However, in the case of Möjsjövik, it is the artificial radionuclide (^{137}Cs) which contributes significantly higher in percent (80%) to the annual effective dose rate than the natural ones (20%).

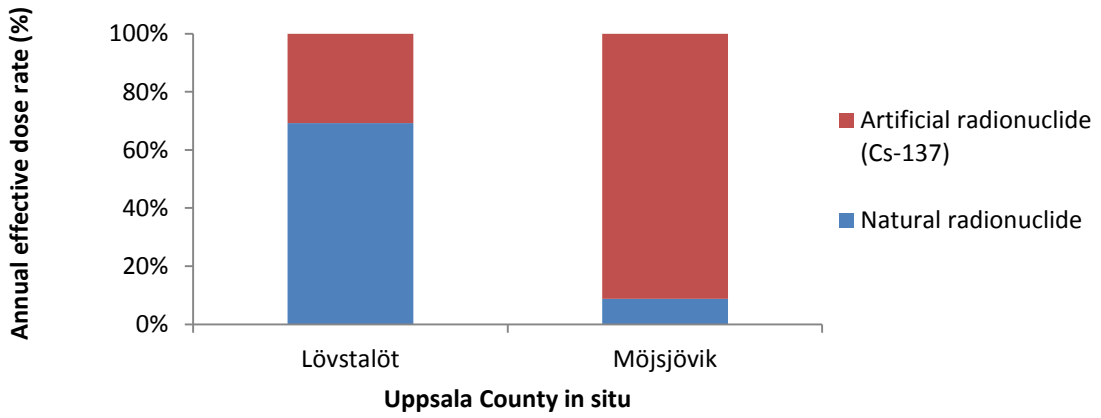

Figure 19 A comparison of percentage contribution of artificial and natural radionuclides to the annual effective dose rates in situ measurement of Jämtland County.

Figure 19 illustrates the in situ analysis of the annual effective dose rate of natural and artificial radionuclides (in air at height of 1 m from the surface) in Lövstalöt and Möjsjövik study areas. In Lövstalöt study area, the percent contribution of natural radionuclides to annual dose rate is almost 70% which is significantly higher than the artificial one (^{137}Cs), 30%. However, in the case of Möjsjövik, it is the artificial radionuclide (^{137}Cs) which contributes significantly higher in percent (90%) to the annual effective dose rate than the natural ones (10%).

6.2 Jämtland County

6.2.1 Activity concentration in soil and grass

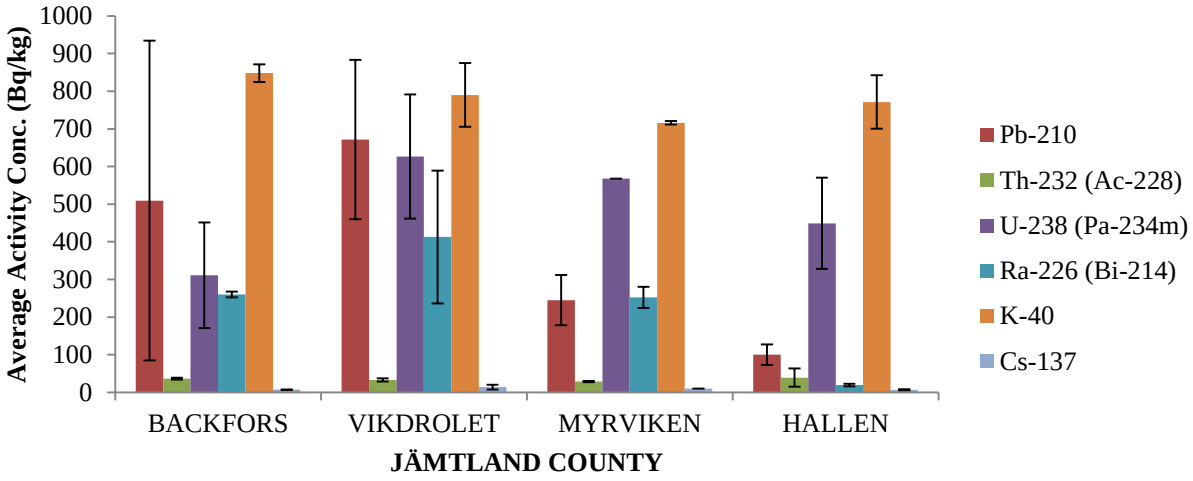

Figure 20 The average activity concentration of radionuclide elements at the soil depth of 5 cm in Jämtland County.

Figure 19 shows the average activity concentration of radionuclide elements in Backfors, Vikdrolet, Myrviken, and Hallen study areas in Jämtland County at soil depth of 5cm. The average activity concentration at this depth was used to calculate transfer factor from soil to grass assuming that the effective root depth of the grass in Jämtland pasture land was 5 cm.

It demonstrates that the average activity concentration of natural radionuclides was significantly higher than that of ^{137}Cs. ^{40}K is the most dominant radionuclide element comparing with the rest radionuclide elements in Jämtland County. In Backfors, ^{40}K was estimated to have the highest average activity concentration compared to the rest of the study areas.

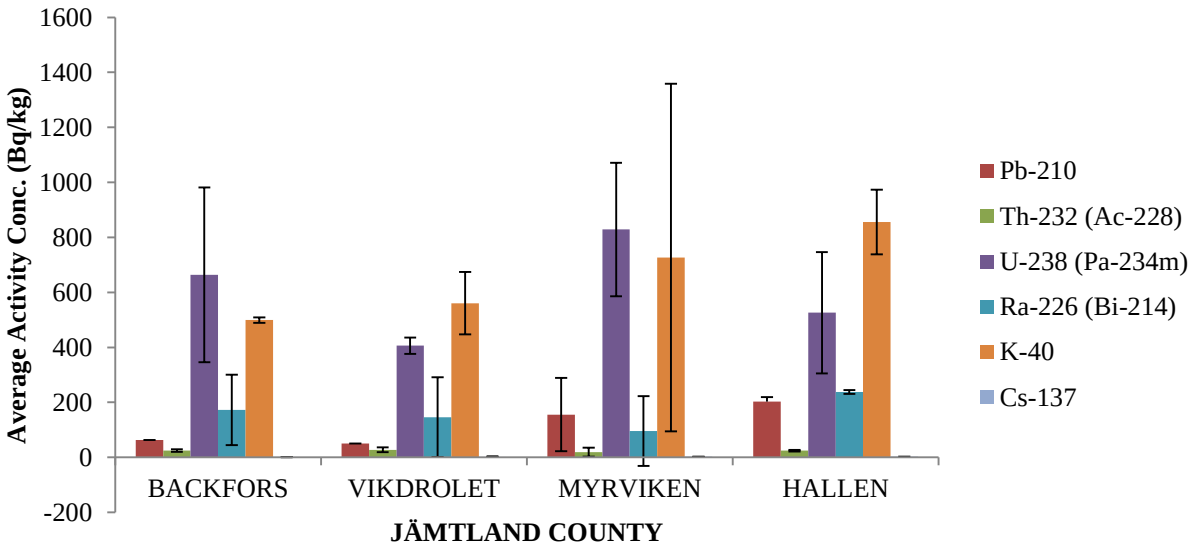

Figure 21 The average activity concentration of radionuclide elements in grass in Jämtland County.

Figure 20 illustrates the average activity concentration of radionuclide elements in grass in Backfors. Vikdrolet and Myrviken study areas in Jämtland County. In all the locations, the highest average activity concentration was observed in ^{238}U and ^{40}K. In Hallen, ^{210}Pb and ^{226}Ra were found to be high comparing with the other study areas.

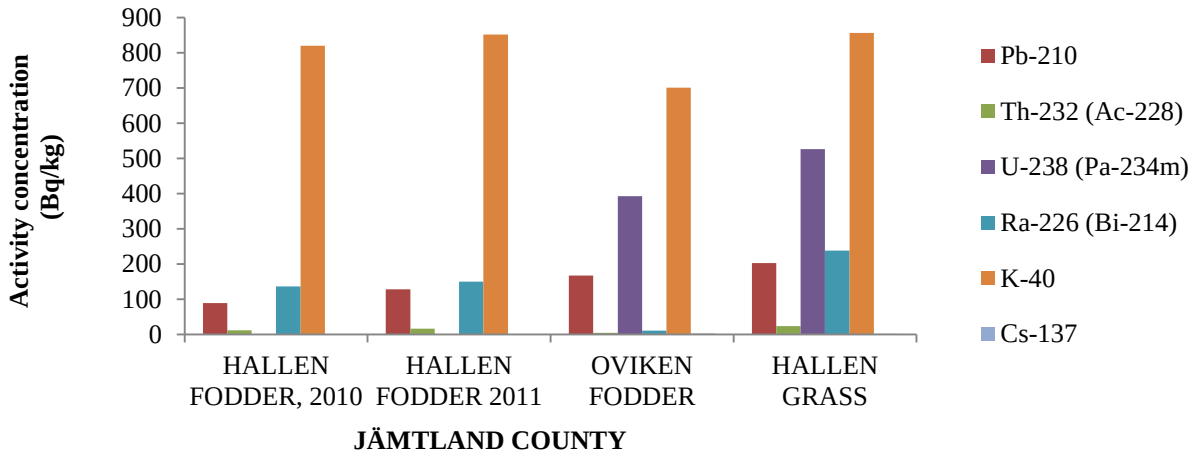

Figure 22 The Activity concentration of radionuclide elements in fodder and grass in Jämtland County.

Figure 21 shows that in Hallen grass, ^{40}K and ^{238}U were the most dominant radionuclide elements. However, in Oviken fodder ^{238}U was estimated the second highest comparing with Hallen fodder 20110 and 2011 which were estimated as zero.

6.2.2 Activity concentration in milk

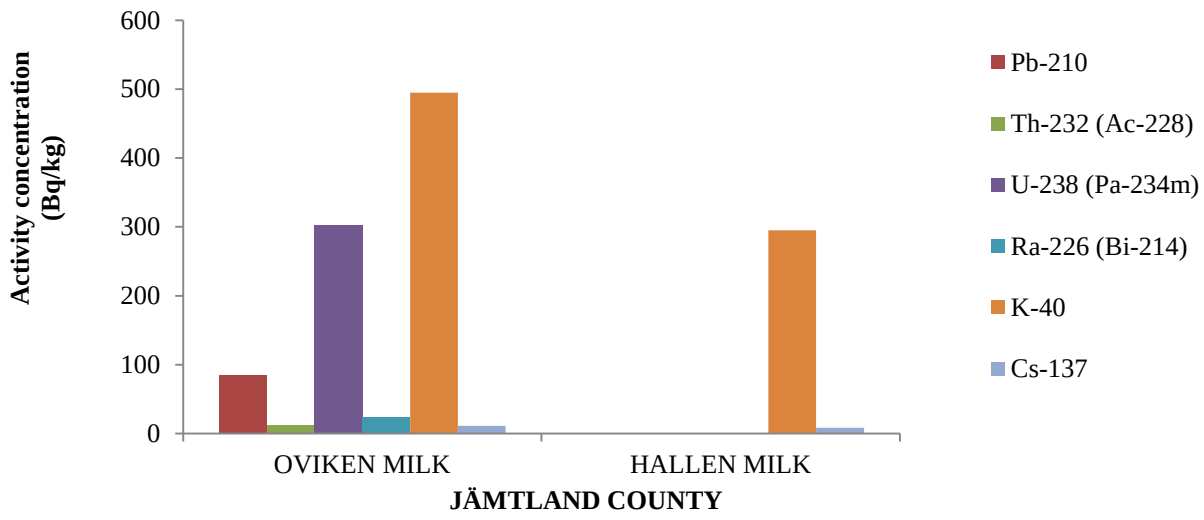

Figure 23 The activity concentration of radionuclide elements in milk Jämtland County.

Figure 22 shows the activity concentration of ^{40}K radionuclide was estimated to be as the most dominant in both Oviken and Hallen milk and ^{137}Cs was estimated to be almost the same in both milks'. However, the activity concentration of ^{40}K was estimated to be higher in Oviken than in Hallen milk. The rest of natural radionuclides (^{238}U, ^{232}Th, ^{210}Pb and ^{226}Ra) were not identified in Hallen milk. In Oviken, these elements were estimated with significant concentration.

6.2.3 Transfer factor

Table 4: The average transfer factor of radionuclide elements in Jämtland County

Radionuclide elements		Study area				
Average Transfer Factor		**Backfors**	**Hallen**	**Myrviken**	**Vikdrolet**	**Oviken**
K-40	Soil to grass	0.59 ± 0.012	1.11±0.06	1.02 ± 0.883	0.71 ± 0.113	-
	Grass to milk		0.35	-	-	-
	Fodder to milk		0.36	-	-	0.706134
Cs-137	Soil to grass	0.04±0.02	0.11±0.041	0.080±0.0354	0.07±0.041	-
	Grass to milk	-	4.1	-	-	-
	Fodder to milk	-	0	-	-	4 362 934
Pb-210	Soil to grass	0	2.12±0.52	0.57 ± 0.508	0	-
	Grass to milk	-	0	-	-	-
	Fodder to milk	-	0	-	-	0
Ra 226 (Bi)	Soil to grass	0.67 ± 0.5	4.19±2.60	0.34 ± 0.443	0.46 ± 0.55	-
	Grass to milk	-	0	-	-	-
	Fodder to milk	-	0	-	-	0
U-238 (pa-234)	Soil to grass	2.57 ± 0.12	0	1.16±0.51	0	-
	Grass to milk	-	0	-	-	-
	Fodder to milk	-	0	-	-	0
Th-232 (Ac-228)	Soil to grass	0.7 ± 0.16	0.622±0.05	0.66 ± 0.585	0.86 ± 0.17	-
	Grass to milk	-	0	-	-	-
	Fodder to milk	-	0	-	-	0

Table 4 shows the average transfer factor of radionuclide elements from soil to grass, grass to milk and fodder to milk in Jämtland County.

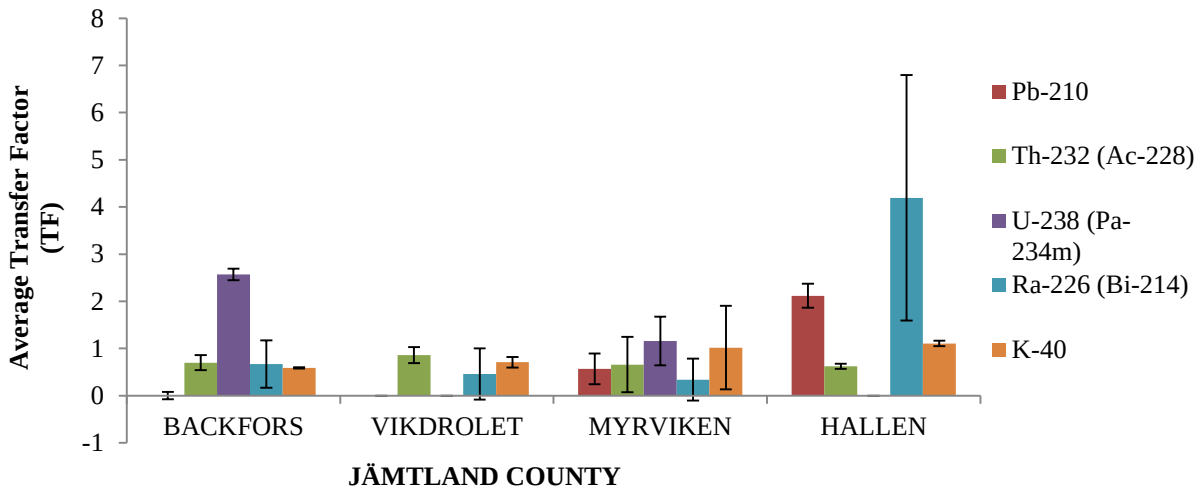

Figure 24 The average transfer factor of natural radionuclides from soil to grass in Jämtland County.

Figure 23 illustrates the average transfer factor of natural radionuclides from soil to grass in Backfors, Vikdrolet, Myrviken, and Hallen study areas. In Backfors, ^{238}U had the highest transfer factor (around 2.8) with low uncertainty and significant difference comparing with

the rest radionuclide elements (< 1), and ^{210}Pb was not identified. In Vikdrolet only ^{232}Th, ^{226}Ra and ^{40} K were identified (transfer factor value of less than 1) with the highest value corresponding to ^{232}Th followed by ^{40}K. However, in Myrviken, all radionuclides were identified, where ^{238}U has the highest transfer factor and ^{40}K was the second highest of all the elements. In Hallen, different results were obtained, where the ^{226}Ra had the highest transfer factor (around 3.7) with high uncertainty and significant difference comparing with ^{210}Pb, ^{40}K and ^{232}Th. The second highest was ^{210}Pb with a transfer factor around 2.

In general, the highest transfer factor was estimated in Hallen by ^{226}Ra, ^{210}Pb, and ^{40}K comparing with other study areas, except ^{238}U in Backfors.

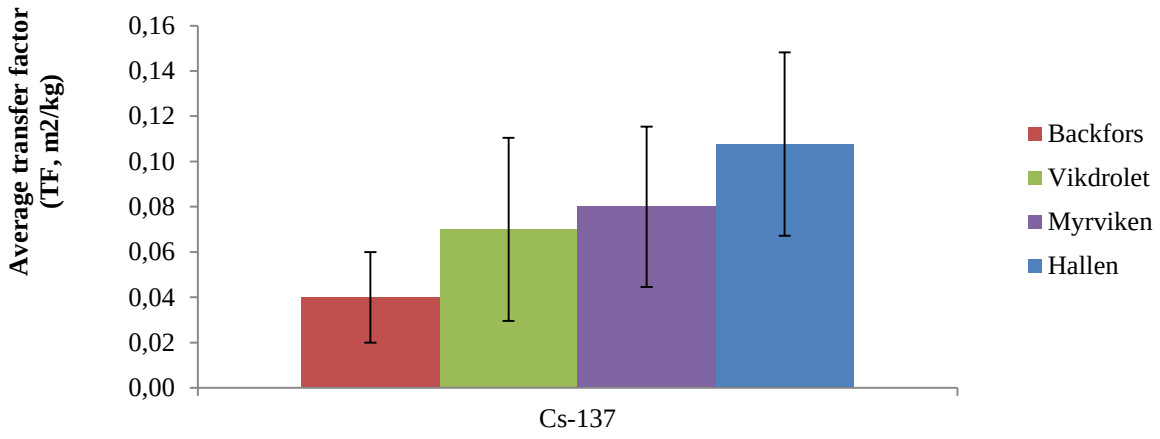

Figure 25 The average transfer factor of Cs-137 from soil to grass in Jämtland County.

Figure 24 shows the transfer factor of ^{137}Cs from soil to grass in Hallen, Myrviken, Vikdrolet and Backfors study areas. The highest value was estimated in Hallen followed sequentially by Myrviken, Vikdrolet and Backfors. From the figure, it could be explained that there was a significant difference of transfer factor of ^{137}Cs between Hallen (the highest 0.15) and Backfors (the lowest, around 0.039).

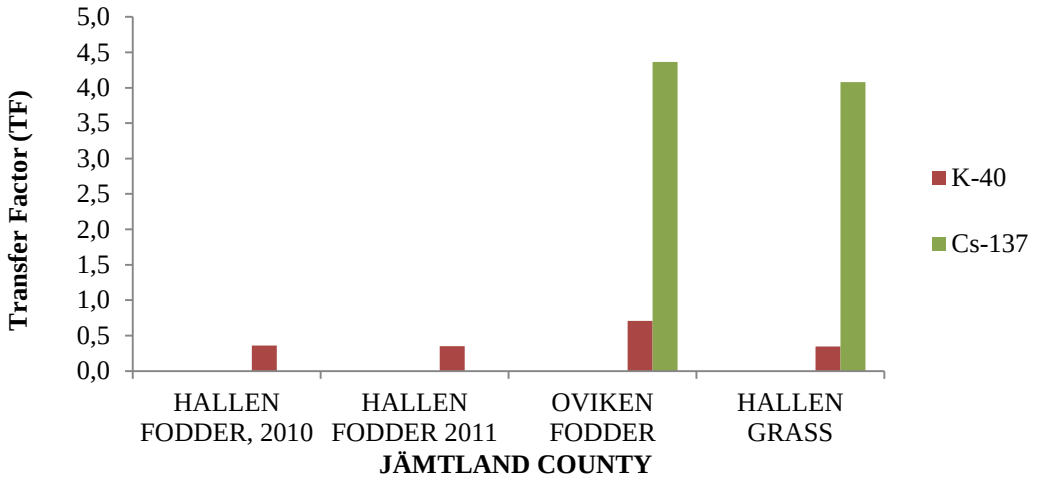

Figure 26 The transfer factor of Radionuclide elements from grass/fodder to milk in Jämtland County.

Figure 25 illustrates the transfer factor of radionuclide elements from grass/fodder to milk in Hallen and Oviken. In Hallen, only ^{40}K was identified and almost the same transfer factor in both 2010 and 2011. However, in Oviken and Hallen, only ^{137}Cs and ^{210}Pb were identified, which had significantly very high transfer factor comparing with Hallen (fodder, 2010 and 2011). Comparing Oviken with Hallen grass, it was Oviken which had the higher transfer factor than Hallen grass. In general, ^{137}Cs had estimated the highest transfer factor with significant difference comparing with ^{210}Pb and the rest in Jämtland County.

6.2.4 Soil profiles

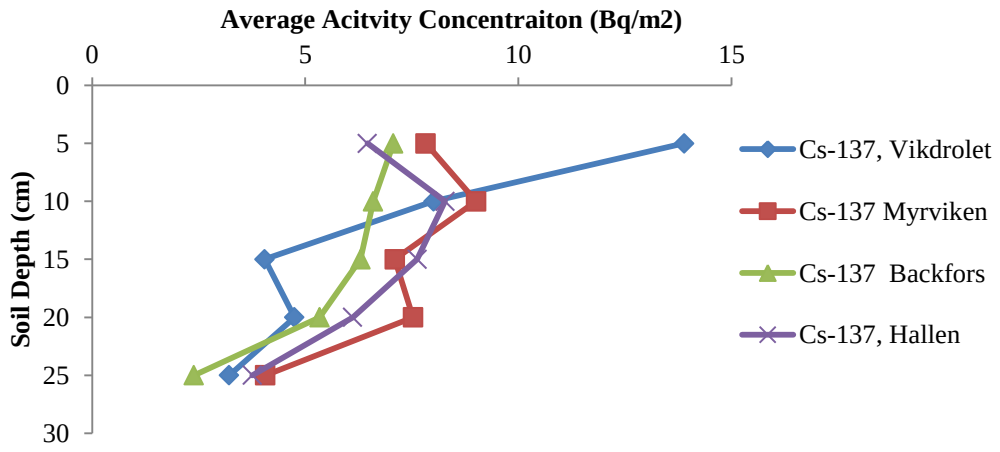

Figure 27 Migration of ^{137} Cs in the soil profile of 25 cm depth in Jämtland County.

Figure 26 illustrates the migration of ^{137}Cs in the soil profile with a depth of 25 cm in Vikdrolet, Myrviken, Backfors and Hallen study areas. In all cases, except in Backfors, the average activity concentration of ^{137}Cs had both an increase and decrease trend or was fluctuated in the profile particularly in Myrviken. This behaviour could be attributed to a vertical downward and upward movement or migration of ^{137}Cs in the profile. However, in Backfors, it could be attributed to only vertical downward migration of ^{137}Cs, since only a decreasing trend in the average activity concentration was observed in the profile.

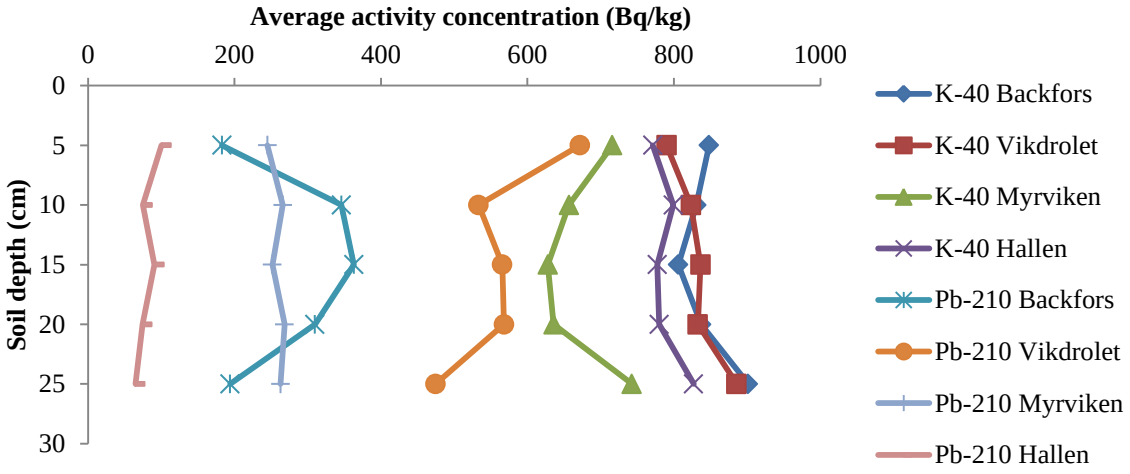

Figure 28 Distribution of ^{210}Pb and ^{40}K in the soil profile of 25 cm depth in Jämtland County.

Figure 27 evaluates the distribution of ^{210}Pb and ^{40}K, in the soil profile within 25 cm depth in Backfors, Vikdrolet, Myrviken and Hallen study areas. The distribution of ^{40}K, was not uniform in the profile of the study areas, since the estimated average activity concentration was not constant. The distribution of ^{210}Pb was not uniform in the profiles of Vikdrolet and Backfors. However, in Myrviken and Hallen, the distribution of ^{210}Pb was fairly uniform in the profile (because of the estimated average activity concentration was slightly different (close to constant) in the profile).

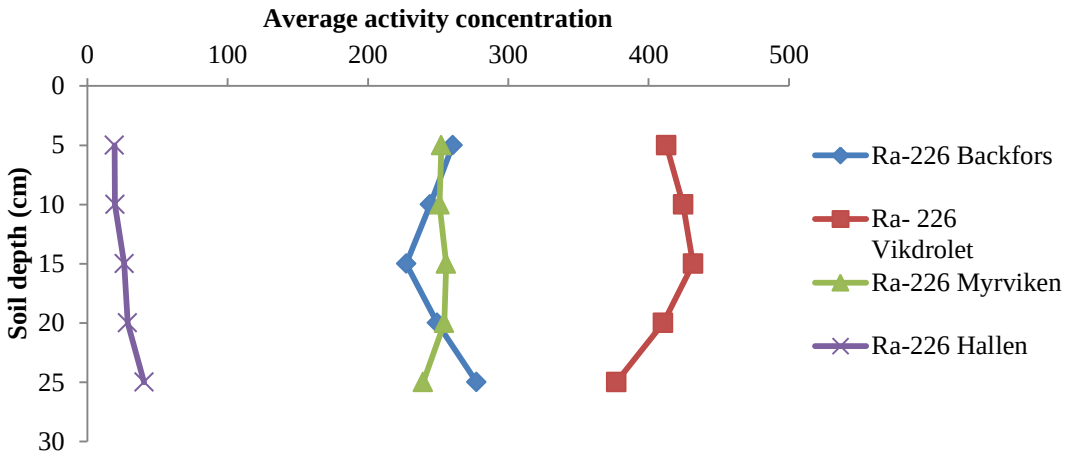

Figure 29: Distribution of Ra-226 in the soil profile of 25 cm depth in Jämtland County.

Figure 28 demonstrates the homogeneity (distribution) of ^{226}Ra in the soil profile with a depth of 25 cm in Backfors, Vikdrolet, Myrviken, and Hallen study areas. In Hallen and Myrviken, ^{226}Ra was almost homogenous throughout the profile (fairly uniform distribution) comparing with the rest. However in Vikdrolet and Backfors, ^{226}Ra was not uniformly distributed as the activity concentration was clearly fluctuating in the profile, which indicates non homogenous distribution.

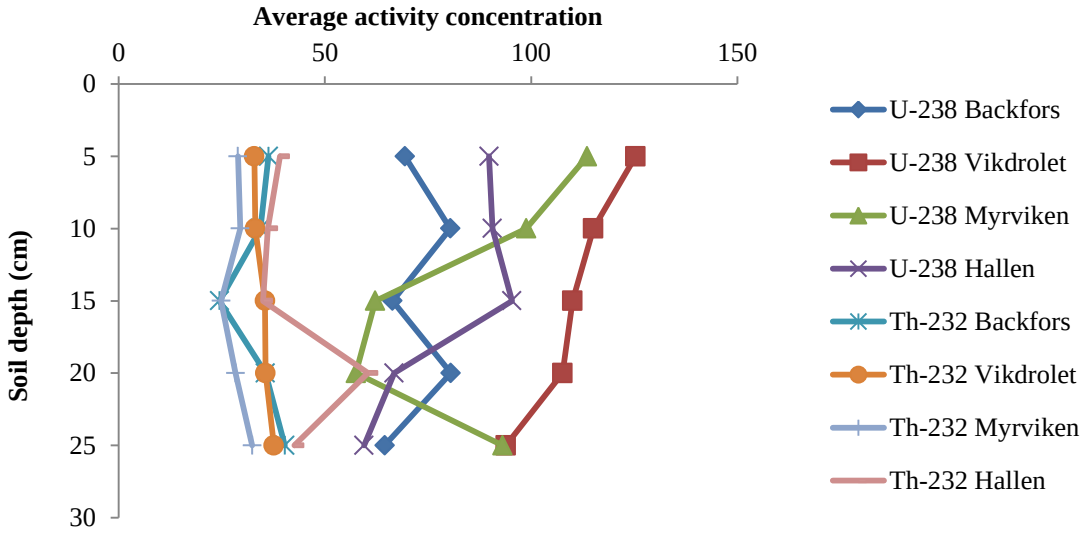

Figure 30 Distribution of ^{238} U and ^{232}Th in the soil profile of 25 cm depth in Jämtland County.

Figure 29 shows the homogeneity or distribution of ^{238}U and ^{232}Th in the soil profile within the depth of 25 cm in Backfors, Vikdrolet, Myrviken, and Hallen. The average activity concentration of ^{238}U was divided by a factor of 5 in order to incorporate both ^{232}Th and ^{238}U

together in the same graph. In the case of ^{238}U, the average activity concentration was not constant in the profile of the study areas. This indicates that the distribution of ^{238}U was non-uniform. A better result was obtained in Vikdrolet, which was slightly uniform. In the case of ^{232}Th, a different distribution trend was observed. In Hallen, it was very homogenous and uniformly distributed in the profile, since its average activity concentration was constant though out the profile. Likewise, in Myrviken, almost a uniform distribution of ^{232}Th was observed. However, in Vikdrolet and Backfors, distribution of ^{232}Th was determined as non-uniform in the profile.

6.2.5 Statistical analysis-Regression and Correlation coefficient

The regression analysis results show that Hallen (93.5%) shows the most statistically significant values followed by Myrviken (89.4%) and Vikdrolet (59.2%). However, Backfors (35.8%), was not statistically significant since its P-value (0.156) was greater than 0.005. This shows that the average activity concentration of radionuclides in grass in all sites, except in Backfors, was dependent on the average concentration of radionuclides in the soil with strong significance in Hallen and Myrviken subsequently.

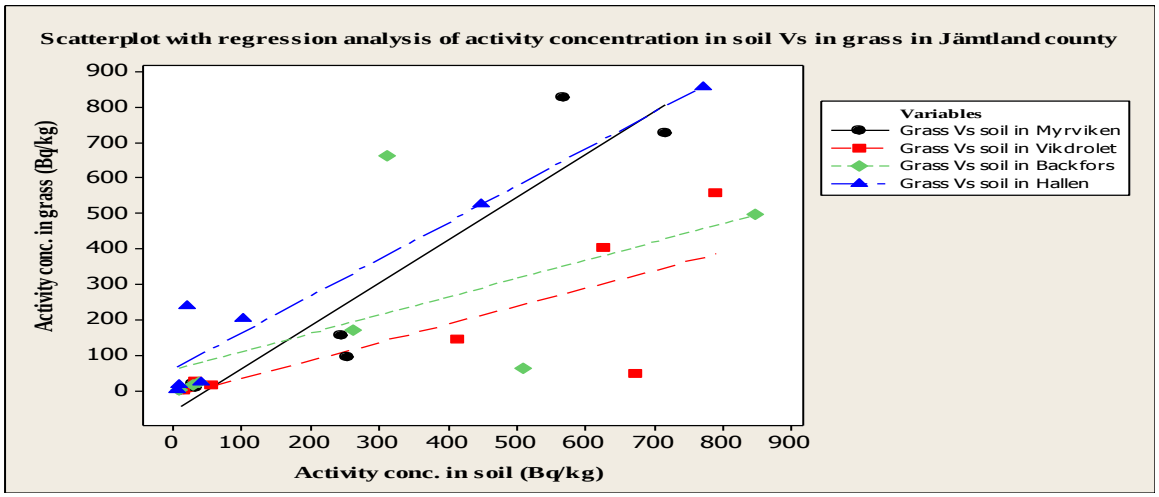

Figure 31 Scatterplot with regression analysis of the average activity concentration in soil versus in grass in Jämtland County.

Figure 30 illustrates scatter plot with regression analysis of average activity concentration of radionuclides in grass versus in soil in Jämtland County. It exhibits that Hallen has the best fitted regression line and closer residual points to linear line comparing with the rest. The second is Myrviken followed by Vikdrolet. However, Backfors has a poorly fitted regression

line with residuals far from the linear fit. Thus, all locations except Backfors, demonstrate that the average activity concentration of radionuclides in grass is dependent on the average activity concentration in soil. Hallen has shown best fitted regression line with strongest dependence of the average activity concentration in grass than soil.

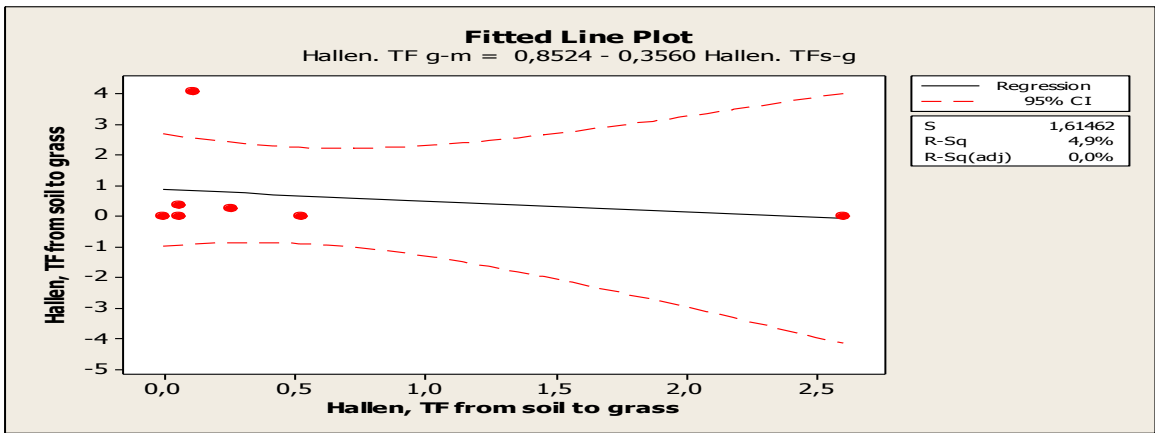

Figure 32 Fitted line plot with regression analysis of the TF from soil to grass versus the TF from grass to milk in Lövstalöt study area, Jämtland County.

Figure 31 shows with confidence interval of 95 %, the obtained R-square and P-value are 0.049 and 0.064, respectively. These explain that the TF from grass to milk was dependent on the TF from soil to grass.

6.2.6 Dose assessment

Table 5: the annual effective dose rate of natural radionuclides and Cs-137, and the total annual effective dose rate in air at the height of 1 m in Jämtland County(in situ and laboratory measurement).

Type of Measurement	Study Area	Absorbed dose rate of natural (nGy/h)	Annual effective natural radionuclide (µSv/y)	Annual Effective Dose rate of Cs-137 (µSv/y)	Total Annual Effective Dose Rate (µSv/y)
Lab.	Hallen	6.5E+01	8.0E+01	2.5E+00	8.2E+01
	Vikdrolet	2.4E+02	3.0E+02	5.4E+00	3.0E+02
	Myrviken	1.6E+02	2.0E+02	3.9E+00	2.1E+02
	Backfors	1.8E+02	2.2E+02	3.0E+00	2.2E+02
In situ	Hallen	7.6E+01	9.3E+01	0.0E+00	9.3E+01
	Vikdrolet	1.9E+02	2.3E+02	6.2E+00	2.4E+02
	Myrviken	1.2E+02	1.4E+02	6.5E+00	1.5E+02
	Backfors	9.3E+01	1.1E+02	0.0E+00	1.1E+02

Table 5 describes the annual effective dose rate of natural radionuclides and ^{137}Cs, and their total (sum) annual effective dose rate in situ and labaratory measurements in Backfors, Hallen, Myrviken, and Vikdrolet study areas. In the case of natural radionucldies, first the annual absorbed dose rate was calculated using equation (8), and then the annual effective dose rate (equation 9). However, in the case of ^{137}Cs, it was directly converted from activity concentration in Bq/m2 in to the annual effective dose rate using the conversion factor . Vikdrolet has the higest total annual effective dose rate of radionuclides both in the labaratory (3.0E+02 µSv/y) and in situ (2.4E+02) respectively. In the laboratory measurement, Backfors presents the second highest dose rate of 2.2E+02 µSv/y followed by Myrviken, 2.1E+02 µSv/y. However, in situ, the second highest dose rate is Myrviken (1.5 E+02 µSv/y) followed by Backfors (1.1E+02 µSv/y). Hallen has the lowest dose rates both in laboratory and in situ, 8.2 E+01 µSv/y and 9.3 E+01 µSv/y respectively. In general, the study shows that the estimated total annual effective dose rate in laboratory is higher than in situ in all study areas except in Hallen.

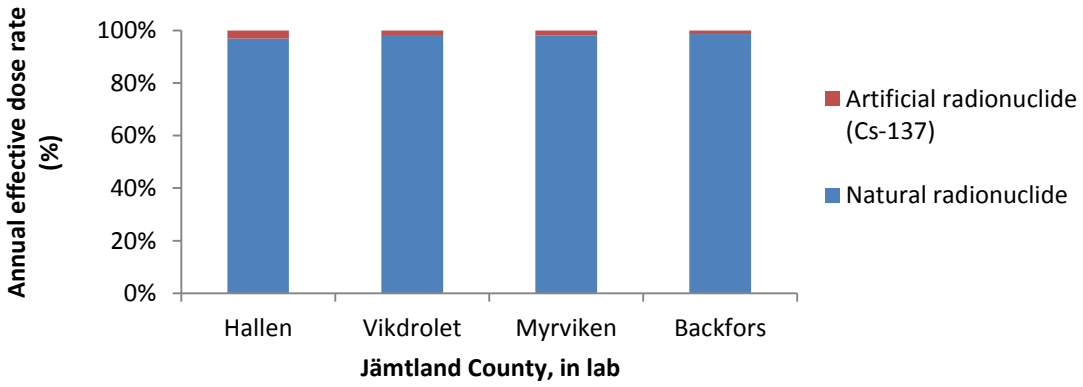

Figure 33 Comparison of the percentage contribution of artificial and natural radionuclides to the annual effective dose rates in laboratory measurement of Jämtland County.

Figure 33 illustrates the laboratory analysis of the annual effective dose rate of radionuclides in percent in Backfors, Hallen, Vikdrolet, and Myrviken study areas. The difference between the percent of annual effective dose rate between artificial and natural radionuclides measured in the laboratory is very significant. In all study areas except in Hallen, the percent contribution of artificial radionuclides to the annual effective dose rate are very small (<5%) in comparison to the natural radionuclides which are greater than 95 %. Hence, the main concern is given to the natural radionuclides in Jämtland County.

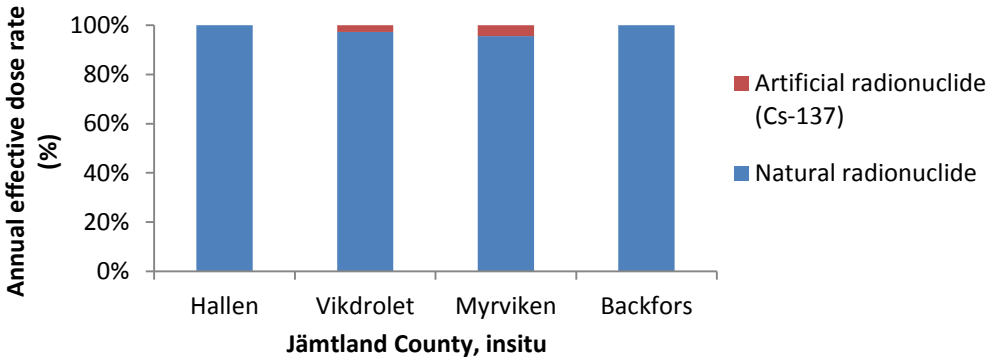

Figure 34 Comparison of the percentage contribution of artificial and natural radionuclides to the annual effective dose rates in situ measurement of Jämtland County.

Figure 33 illustrates the in situ analysis of the annual effective dose rate of radionuclides in percent in Backfors, Hallen, Vikdrolet, and Myrviken study areas. The difference in the percent of annual effective dose rate of artificial and natural radionuclides measured in the laboratory is very significant. In Vikdrolet and Myrviken study areas, the percent contribution of artificial radionuclides to the annual effective dose rate are very small (<5%) in comparison to the natural radionuclides which are greater than 95 %. However, in the case of Hallen and Backfors, there is no any contribution of artificial radionuclides to the annual effective dose rate. On the other hand, the contribution of natural radionuclides to the annual effective dose rate almost is between 95-100%. Hence, even in the situ measurement, the main concerns should be given to the natural radionuclides.

7. Discussion

7.1 Soil samples

In this study, the activity concentration of ^{40}K, ^{238}U, ^{137}Cs, ^{232}Th, ^{210}Pb and ^{226}Ra in the soil profile at the depth of 25 cm (5 layers with each 5 cm depth) range between 38.9 – 1770 Bq/kg, 82.6 – 1830 Bq/kg, 0 – 1110 Bq/kg, 3.35 – 101 Bq/kg, 0 – 897 Bq/kg and 10.6 – 681 Bq/kg, respectively. The highest average activity concentration of ^{40}K, ^{238}U, ^{137}Cs, ^{232}Th and ^{226}Ra were found in Lövstalöt, Möjsjövik and Vikdrolet respectively (see table 13, appendix 4).

Table 6: Comparison of average activity concentration (Bq/kg) of Uppsala and Jämtland with global data

Places	K-40	Cs-137	Pb-210	Ra 226 (Bi)	U-238 (pa-234)	Th-232(Ac 228)	Reference
Uppsala	269.37 - 997.2	63.25-753.67	84.42-87.85	61.13-112.22	485.73-873.6	38.98-60.44	Current study
Jämtland	675.8-844.67	5.29-9.47	80.97-562.67	22.93-411.27	330.4-533.73	28.79-42.66	Current study
Global UNSCEAR (2000)	400 -	- -	- -	35 -	- 35	30 -	(Dai L. et al., 2006 Papp Z. et al., 2001
China	584	-	-	37.6	-	54.6	Dai L. et al., 2006
Ajka	329	17.4		136	130	25.5	Dai L. et al., 2007
M.Vaccaro	-	-	23.3± 3	-	-	-	Battaglia et al., 1988
C. Semes	-	-	1173± 5	-	-	-	Battaglia et al., 1989
Tuturano	-	-	31.63± 3.4	-	-	-	Battaglia et al., 1990
M.badessa	-	-	19.83± 7.2	-	-	-	Battaglia et al., 1991
C.Trio	-	-	634± 10	-	-	-	Battaglia et al., 1992
Case Bianchet	-	-	6023± 9	-	-	-	Battaglia et al., 1993
Algeria	93 – 412	15 – 35	-	-	5 – 27	7 – 27	Noureddine et al., 1997
Bangladesh	200 – 772	0.40 – 3.88	-	-	15.2 – 94.0	28.3 – 129.0	Chowdhury et al., 1999
Kuwait	4 – 497	0.1 – 9.9	-	-	1.8 – 28.2	1.5 – 16.3	Bou-Rabee, 1997
Norway	31 – 564 (283)	0.9 – 146.0 (34.8)	-	-	12 – 137 (43.3)	4 – 52 (21.1)	Dowdall et al., 2003

From the above table 6, it can be described that the average (range) activity concentration of ^{40}K, ^{238}U, ^{137}Cs, ^{232}Th, and ^{226}Ra in the soil profile (25 cm depth) of the study areas were

higher than that of the values recommended by UNSCEAR (2000) and the findings of studies made in Chines, Italy, Algeria, Bangladesh, Norway and Ajka soils.

Table 7: Cs-137 deposition

Study area	CS-137 deposition (Bq/m2)
Lövstalöt	1.39
Möjsjövik,	5.03
Vikdrolet	0.24
Myrviken	0.29
Hallen	0.18
Backfors	0.15
Hille (Rosén et al.1998)	178.5
Trödje (Rosén et al. 1998)	184

Since ^{137}Cs was anthropogenic deposited radionuclide, it was estimated in Bq/kg and converted into Bq/m^2 considering the deposited area of the soil. The average activity concentration of ^{137}Cs in the soil profile (0 – 25 cm) study areas were lower than the results found by Rosén et al. (1998) studying on the distribution of ^{137}Cs at soil profile (0 – 25 cm) in the soil horizons of Hille (1998) and in Trödje (1990) sites. The reason for this difference could be due to the half-life decay of ^{137}Cs, is around 20 years when Chernobyl accident was occurred 25 years ago.

In Uppsala County, the average activity concentration of ^{238}U, ^{137}Cs and ^{226}Ra in Möjsjövik soil surface at 5 cm depth are higher than in Lövstalöt soil. But ^{40}K is higher in Lövstalöt than in Möjsjövik. This is due to the fact that the Möjsjövik soil was sandy loam and very rich in organic matter, where these radionuclides were poorly bounded in the soil matrix (particles). Because ^{137}Cs in Möjsjövik soil was particularly available in large amount (see table 11, appendix 2), mobile and found in the soil solution in large concentration. It is more easily detected and available to be up taken by plant roots compared to the soil in Lövstalöt. Another reason could be that there was significantly higher deposition of ^{137}Cs from Chernobyl accident in Möjsjövik than in Lövstalöt. In the case of Lövstalöt, the soil is clay loamy and very rich in clay particles. These clay particles bound ^{238}U, ^{137}Cs and ^{226}Ra more effectively compared to Möjsjövik (see table 8, appendix 2). The activity concentration of ^{137}Cs was lower due to the fact that it was strongly bound to clay particles. It was found in soil solution in lower concentration to be easily detected and available for plants roots (see

table 10, appendix 1). In addition, ^{40}K had the highest activity concentration of all the concerned radionuclides in Lövstalöt soil. This was mainly due to the fact that soil was slightly acidic (pH 5.06), which caused the essential elements (such as K) to be available in small amount in soil (See table 10, appendix 1).

In Jämtland, the soil has higher activity concentration of natural radionuclides such as ^{40}K, ^{238}U, ^{210}Pb and ^{226}Ra than ^{137}Cs. This could be attributed to the fact that the soil has relatively higher content of clay particles but lower amount of organic matter, in which the ^{137}Cs was strongly bounded in the clay particles or silicates (See table 7, Appendix 1). It was also reported by fawaris and Johanson (1995) that high amount of organic matter in soil (peat soil) reduced ^{137}Cs mobility (El-Reefy et al., 2006). In contrast, results reported by Szerbin et al. (1999) indicated that ^{137}Cs is immobilized rapidly in soils which contain small amount of organic matter (El-Reefy et al., 2006).

In general, the study estimated the average activity concentration of ^{40}K , ^{238}U, and ^{232}Th at the soil depth of 5cm in the study areas range 689.9 – 969 Bq7kg, 311 – 1513.33 Bq/kg and 28.77 – 60.33 Bq/kg (see table 8, appendix 2). These are higher than those calculated in Istanbul surface soil samples which are 342 Bq/kg (^{40}K), 21 Bq/kg (^{238}U) and 37 Bq/kg (^{232}Th) (Karahan et al., 1999).

The findings of the current study explain that the soil in Uppsala County had higher ^{137}Cs activity concentration than Jämtland study areas due to larger effect of deposition from Chernobyl accident. In Jämtland ^{137}Cs was available in a very low activity concentration compared to the concerned natural radionuclides. This is due to geological background of natural radionuclides in Jämtland County. In most cases, the current study and Dowdall M. et al. (2003) demonstrated that the activity concentrations of ^{137}Cs was largely lower than those of ^{40}K, ^{238}U, ^{232}Th, ^{210}Pb and ^{226}Ra. The highest activity concentration of ^{137}Cs was found at the soil surface (depth of 5 cm layer) compared to some natural radionuclides more or less uniformly distributed in the study areas**.** This proves that ^{137}Cs was solely a source of anthropogenic deposition (Dowdall M et al., 2003).

7.2 Grass samples

In Uppsala County, the estimated activity concentration of ^{137}Cs in grass was significantly higher in Möjsjövik than in Lövstalöt study area. In addition, the estimated Transfer factor from soil to grass in Möjsjövik (1.25) was 100 times higher than in Lövstalöt (0.01). These could be attributed to the fact that Möjsjövik soil was organic soil (no clay particles) whereby ^{137}Cs was easily available for plant (grass) roots (see table 14 and 15, appendix 3). However, in Lövstalöt, the transfer factor (TF) was almost zero due to the high content of clay particles which strongly bound ^{137}Cs and made less available for plant roots. Among the alkali elements, ^{137}Cs is strongly bound to clay minerals due to its small hydrated radius and strongly accumulated by plant species (Rosén et al., 2009 and Román, 2006).

With regard to Jämtland, the highest activity concentration of ^{137}Cs in grass was estimated in Myrviken followed by Hallen. This was due to higher activity concentration of ^{137}Cs in Myrviken soil than in Hallen. However, the estimated transfer factor from soil to grass was higher in Hallen (0.1) than in Myrviken (0.08). This could be attributed to the fact that Hallen soil was acidic and had higher percentage of sand (sandy soil) and at the same time lower content of essential elements (such as Ca, Mg, Rb, K, etc.) for plant growth than Myrviken. The comparison between Counties explained that the highest average transfer factor of ^{137}Cs from soil to grass was estimated in Möjsjövik study area followed by Hallen. But the least was estimated in Lövstalöt due to high percentage of clay but very low sand and organic matter content in the soil. This describes that transfer factor of radionuclides from soil to grass is largely affected by soil properties and soil texture.

According to Rosén et al. (1998), studies on the soil-plant transfer factor were carried out at different sites of grass lands in central and northern Sweden and significant differences were observed. The sites with the highest transfer factor were rich in gravelly sandy loam soils and organic matter rich surface horizons, intermediate in the organic soils and the lowest in soils from the river and coastal areas (Rosén et al., 1998).

In Jämtland County, the average transfer factor (TF) of natural radionuclides from soil to grass were between 0.00 (^{210}Pb and ^{238}U) and 4.19 (^{226}Ra) in Backfors and Hallen respectively. The highest was estimated in Hallen ^{226}Ra (4.19), followed by ^{238}U (2.57) in Backfors, ^{210}Pb with TF of 2.12 in Hallen, ^{238}U with 1.16 in Myrviken, and K-40 with 1.11 in

Hallen. However, in Uppsala the estimated average transfer factor of natural radionuclides was between 0 and 2.72 (^{40}K and ^{210}Pb respectively) where both obtained in Möjsjövik study area. Hence, the estimated average transfer factor of natural radionuclides was higher in Jämtland County than in Uppsala County. A study on the transfer of radionuclides from soil to grass was carried out in northern Taiwan where the obtained results for ^{40}K and ^{137}Cs from soil to grass were 1.56 0.55 (0.67 – 2.84) and 0.28 0.19 (range 0.006 – 0.63) respectively (Wang et al., 1996).

In Uppsala County, both study areas (Lövstalöt and Möjsjövik) showed that the average activity concentration of the concerned radionuclides in grass were strongly dependent and positively correlated with that in the soil based on the obtained P-value and R-square value which were also statistically significant to be interpreted. In Jämtland, however, the statistical analysis showed that the average activity concentration of radionuclides (^{238}U, ^{210}Pb, ^{226}Ra, ^{40}K and ^{137}Cs) in grass in all study areas except Backfors were positively correlated with that of the soil and statistically significant to be interpreted based on the data (see table 5). On the other hand, the activity concentrations of the radionuclides in grass in all study areas except in Backfors, were dependent on that of soil as shown in fitted regression linear lines (see figure 28). Backfors had poorly fitted points to the regression linear line having many residuals far away from the line. But the best fitted points to regression linear line was obtained in Hallen study area.

7.3 Milk samples

In the study, different results were obtained concerning the transfer factor from grass to milk. In Uppsala County, a study on the activity concentration of radionuclide elements in milk was carried out in Lövstalöt study area. It was estimated that activity concentration of ^{238}U was the highest of all the nuclide elements. ^{40}K was the second highest compared to the rest. Both ^{40}K and ^{238}U activity concentrations were significantly higher than the rest (see figure 7). And, the transfer factor of ^{137}Cs from grass to milk was estimated to be the highest with significant difference (3.58) from the rest of nuclide elements. This was followed by ^{238}U (0.87) as second and ^{40}K (0.75), as third highest in the study areas (see figure 9).

In Jämtland County, a study on the activity concentration of radionuclide elements in milk were carried out in Hallen and Oviken study areas. In Oviken, all concerned radionuclides

were present in the milk with ^{40}K having the highest concentration followed by ^{238}U. But in Hallen, only ^{40}K was present with comparatively lower concentration than in Oviken. With regard to ^{137}Cs, almost the same concentrations were estimated in both study areas (see figure 20). In addition, the transfer factor of radionuclide elements from grass to milk was estimated in Hallen fodder, Hallen grass and Oviken fodder. According to figure 23, ^{137}Cs had very high transfer factor from grass to milk in Oviken fodder (4.50) and Hallen grass (4.0) compared to the Hallen fodder 20110 (0) and 2011 (0). But ^{40}K had almost the same transfer factor around 0.35 in all study areas except in Oviken slightly higher, 0.51. However, the transfer factor of ^{137}Cs from grass to milk was significantly higher than ^{40}K in both Oviken fodder and Hallen grass (see figure 23).

In Uppsala, the P-value (0.085 > 0.05) and R-square (48%) explained that it was not statistically significant whether the transfer factor from grass to milk was dependent on the transfer factor from soil to grass. Thus it was difficult to define their relationship based on findings (see figure 15). While in Jämtland, based on the obtained P-value (0.064 > 0.05) and R-square (4.9%) at 95% confidence interval, the relationship between the transfer factor from grass to milk and from soil to grass were not statistically significant (see figure 29). Hence, it was very difficult to explain the dependence (the relationship) of the TF from grass to milk on the TF from soil to grass in the study areas. It is also not possible to draw conclusions from this fitted line plot and the trend (see figure 29). This could be due to the difference in the chemical and biological behavior of radionuclides at different components of an ecosystem and food chain/food web of living organisms.

7.4 Soil profiles

In Uppsala County, the migration of ^{137}Cs in soil profile had shown different trend in the study areas. In Lövstalöt, there was vertical downward movement of ^{137}Cs in the soil profile but in Möjsjövik, both vertical upward and downward proved by the fluctuation of activity concentration in the soil profile (See figure 10 and appendix 4). In Möjsjövik, the distribution of natural radionuclides (^{210}Pb, ^{40}K, ^{238}U, ^{232}Th, and ^{226}Ra) was significantly not homogenous in the soil profile. However, in Lövstalöt all radionuclides except ^{232}Th, and ^{40}K were not homogenously distributed in the soil profile. The non-homogenous distribution of ^{210}Pb in both study areas advocates that there was disturbing forces in the areas such as soil erosion

etc. In Jämtland County, the migration ^{137}Cs in the soil profile were both vertical upward and downward in all study areas except vertical downward in Backfors (see figure 24). In case of homogeneity study, ^{210}Pb and ^{232}Th in Vikdrolet, ^{226}Ra in Hallen, and ^{210}Pb in Myrviken were almost homogenously distributed in the profile. In the rest, the distribution was not homogenous (see figure 25, 26 and 27). Specifically, the non-uniform distribution of ^{210}Pb indicates that there was perhaps the occurrence of external forces in the area such as soil erosion.

7.5 Dose assessment

In Uppsala County, the total annual effective dose assessment above the soil surface was found to be higher in Möjsjövik than in Lövstalöt (6.3E+02 and 1.7E+02 μSv/y respectively).In the case of in situ measurement, it was also higher in Möjsjövik than in Lövstalöt (2.6E+02 μSv/y and 1.4E+01 μSv/y respectively (see table 3). The difference in total effective dose rate in both study areas between measurements could be due to the presence of different environmental conditions and detectors properties. The effective dose rate of ^{137}Cs was found to be higher in Möjsjövik (4.8E+02 μSv/y and 2.E+02 μSv/y) than in Lövstalöt (3.8E+01 and 4.3E+01 μSv/y) in both measurements. But the contribution of ^{137}Cs to the total annual effective dose rate was very small. However, this does not mean that the effect of ^{137}Cs to human being and ecosystem was very small, since human beings and ecosystem can be affected even at low dose rate of radionuclides.

In Jämtland County, the total annual effective dose rate at the surface measured in the laboratory and situ were 3.0 E+02 and 2.4E+02 μSv/y in Vikdrolet, 2.1E+02 and 1.5E+02 μSv/y in Myrviken, 2.2E+02 and 1.1E+02 μSv/y in Backfors and 8.2E+01 and 9.3E+01 μSv/y in Hallen. In Hallen study area, higher total annual effective dose rate was estimated in situ than in the laboratory. And, the highest total annual effective dose rate was obtained in Vikdrolet in both measurements. In general, there is significant difference in the total annual dose rate of the study areas estimated by the measurements. In laboratory measurement, the highest total annual effective dose rate was estimated in Uppsala County (i.e. Möjsjövik with 6.3.E+02 μSv/y) but in situ measurement, the highest was obtained in Jämtland (i.e. Vikdrolet with 3.0E+2 μSv/y) (see table 3 and 5). A similar study on ploughed arable land in central Sudan found that the absorbed dose rate in air in central Sudan calculated on the basis of the

secular equilibrium in both the ^{238} U and ^{232}Th series was slightly higher than the one estimated directly from the photo peaks of individual gamma-emitting radionuclides (Sam et al., 2007).

Table 8: Absorbed dose rate measurements in the lab

Study area	Absorbed dose rate (nGy/h) lab measurement	Reference
Uppsala	1.1E+02 - 1.2+02	Current study
Jämtland	6.5E+01 – 2.4E+02	Current study
global primordial radiation	59	Lu et al., 2011
China	62	Lu et al., 2011
Shaanix	63	Lu et al., 2011
Boaoji (China)	86.6±3.4	Lu et al., 2011
Sinnar (Central Sudan)	39±7	Sam et al., 2007

Table 8 described that the absorbed dose rate of study areas in both Counties were higher than that of recommended by UNSCEAR and the findings of studies carried out in China and central Sudan.

Table 9: Effective dose rate measurements in the lab and in-situ

	Natural radionuclides		
	Study area	Effective dose rate (µSv/y)	Reference
Lab	Uppsala (Lab)	1E+02-1E+03	Current study
	Jämtland	8E+01-4E+02	Current study
	Boaoji (China)	106±4	Lu et al., 2011
	central Sudan	47.8 ± 6	Sam et al.,2007
	Istanbul	60	Karahan et al., 1999
In-situ	Uppsala	3E+01-1E+02	Current study
	Jämtland	1E+02-4E+02	Current study
	Istanbul	80	Karahan et al., 1999

Table 9 compares the effective dose rate of Uppsala and Jämtland Counties with that of China, Istanbul and central Sudan in both in laboratory and in situ measurements. It further describes that the study dose rate was higher than the findings of other studies obtained in the above mentioned countries.

In general, the percent contribution of artificial radionuclide (^{137}Cs) to the total annual effective dose rate is significantly very high in Uppsala County comparing with Jämtland County. In contrast, in the case of Jämtland, its contribution is very small and not significant in both measurements.

8. Conclusion

- The findings of the study showed that the activity concentration of the natural radionuclides (^{210}Pb, ^{40}K, ^{232}Th, ^{226}Ra and ^{238}U) and ^{137}Cs of the study areas were higher than that of world average according to UNSCEAR.
- The concentration of ^{137}Cs in a soil was higher in the study areas of Uppsala Counties than that of Jämtland due to higher deposition of anthropogenic nuclides (^{137}Cs) in the County.
- The concentration of natural radionuclides (^{210}Pb, ^{40}K, ^{232}Th, ^{226}Ra and ^{238}U) in a soil were found to be higher in Jämtland than in Uppsala County because of the geological background of the area.
- ^{137}Cs had the highest transfer factor from soil to grass than natural radionuclides in Uppsala study areas. In Jämtland, the transfer factor (TF) of natural radionuclides was found to be higher than that of ^{137}Cs. ^{137}Cs had the highest transfer factor from grass to milk of all the concerned radionuclides which was obtained in Oviken fodder and Hallen grass.
- The activity concentration of radionuclides in soil was positively correlated with the activity concentration of radionuclides in grass in all study areas except in Backfors.
- The migration of ^{137}Cs in the both Counties was discovered to be both vertical downward and upward even lateral movement was possible depending on the chemical properties of ^{137}Cs, the properties and physical and chemical processes of the soil in the study area.
- The homogeneity of the natural radionuclides (^{210}Pb, ^{40}K, ^{232}Th, ^{226}Ra and ^{238}U) in the study areas, could be determined by their activity concentration throughout the soil profile, which could also suggest the occurrence of external forces.
- Both in laboratory and in situ measurements, Uppsala had the highest total annual effective dose rate comparing with Jämtland (Möjsjövik).
- The study estimated higher absorbed dose rate of natural radionuclides and total annual dose rate than that of China, central Sudan and UNSCEAR.
- In Uppsala County, artificial radionuclide ((^{137}Cs) from Chernobyl accident has shown a very significant contribution to the total annual effective dose rate. However, in Jämtland, its contribution is not significant. Therefore, in Jämtland County, much

concern is given to natural radionuclides but in Uppsala, both artificial (^{137}Cs) and natural radionuclides are very important to be considered for dose rate assessment.

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Appendices

Appendix 1- Soil texture and soil properties of study areas.

Table 10 Soil texture and soil properties of the study areas in Jämtland and Uppsala Counties

	Percentage (%)							Mg/100g					
Study Area	**clay**	**silt**	**loam**	**sand**	**organic matter**	**Tot-C**	**Tot-N**	**pH**	**P-AL**	**K-AL**	**Ca-AL**	**Mg-AL**	**S-AL**
Myrviken	25	27	20	29	21	9.5	0.9	6.23	4.6	10.9	591.8	25.5	5
Hallen	18	20	17	44	12	4.3	0.4	6.11	6.9	8	312.2	17.5	2.3
Backfors	28	38	26	8	8	3.1	0.3	5.82	1.1	4.2	196.9	9.3	0.7
Vikdrolet	23	23	25	29	10	4.1	0.4	5.66	1.5	8.8	239.1	7.6	1.2
Möjsvjövik	-	-	-	-	82	44.4	3	5.45	3.9	36.2	959.6	32.4	8.6
Lövstalöt	31	30	35	4	8	3.3	0.3	5.06	2.1	8.5	119.2	9.7	2.1

Appendix 2 –Average activity concentration of radionuclides at 5cm soil depth, grass, fodder and milk

Table 11: The average activity concentration of radionuclide elements in soil at the depth of 5 cm (Bq/kg) of the study areas, Uppsala and Jämtland Counties

Radionuclide elements	**Backfors**	**Vikdrolet**	**Myrviken**	**Hallen**	**Lövstalöt**	**Möjsjövik**
Pb-210	509,5±424,97	671,67±211,67	245±66,55	100,3±27,33	82,33±27,58	89,6±0
Th-232	36,33±2,65	32,87±4,31	28,77±1,38	39,1±5,96	60,33±13,44	51,87±18.56
U-238	311±140.01	626.5±164.76	568±0.00	449±121.25	635.00	1513.33±348.47
Ra-226	260.33±7.64	412.67±176.73	252.33±28.04	19.4±3.36	68.2±2.31	190±80.61
K-40	848±23.26	790±84.55	716±4.58	771.3±71.28	969±90.57	689.9±937.59
Cs-137	7.07±0.33	13.89±6.00	10.3±0.14	6.46±1.89	145.33±18.56	865±226.78
Cs-137 (Bq/m^2)	210.83±17.08	384.9±108.52	282±35.36	180.8±47.5	2700.6±218.33	34611.97±2789.36

Table 12 The average activity concentration of radionuclide elements in grass (Bq/kg) of the study areas, Uppsala and Jämtland Counties.

Radionuclides	Backfors	Vikdrolet	Myrviken	Hallen	Lövstalöt	Möjsjövik
U-235	16,9±2,01	17,35±0,78	8,61±9,75	16,1±0,74	11,07±0,61	10,03±1,24
Pb-210	63±0,00	49,8±0,00	155,47±133,02	203±16,09	154±9,17	130±0
Th-232	24,25±4,88	27,05±8,84	18,49±16,14	24,2±2,62	20,83±6,69	16,7±0
U-238	664±318,02	406±29,70	828,5±242,54	526±220,38	430,5±19,09	571±0
Ra-226	172,43±127,52	146,2±145,38	95,08±126,96	238±6,93	90,75±105,01	92,45±89,87
K-40	499±9,64	560,33±113,65	726,77±632,11	855,7±118,00	684±149,17	418±133,63
Cs-137	1±0,00	1,82±1,51	3±0,00	2,11±0,33	2,98±1,72	723,67±361,89
Cs-137 (Bq/m^2)	9±0,00	16,52±15,01	21±7,07	18,2±1,44	30,67±14,01	9115±4449,53

Table 13 The activity concentration of radionuclide elements in fodder, grass and milk (Bq/kg) of the study areas, Uppsala and Jämtland Counties.

Radionuclide elements	Hallen fodder (2010)	Hallen fodder (2011)	Hallen grass	Oviken fodder	Lövstalöt grass	Hallen milk	Oviken milk	Lövstalöt milk
U-235	9,51	11,20	16,07	1,29	11,01±0,61	4,15	5,13	6,78±0,10
Pb-210	89,00	128,00	203,00	167,00	154±9,17	0,00	84,50	30,7±27,65
Th-232	12,00	16,80	24,17	4,91	20,8±6,69	0,00	12,90	8,695±4,11
U-238	0,00	0,00	526,00	393,00	430,5±19,09	0,00	303,00	713±398,8
Ra-226	136,00	150,00	238,00	10,70	90,75±105,01	0,00	24,90	35,7±11,78
K-40	820,00	852,00	855,67	701,00	684±149,17	295,00	495,00	516±32,08
Cs-137	0,00	0,00	2,11	2,59	2,98±0,41	8,61	11,30	10,66±6,32
Cs-137 (Bq/m^2)	0,00	0,00	18,18	40,47	44,65±10,00	150,00	164,54	103,6±58,6

Appendix 3- Average activity concentration of radionuclides in soil profile of study areas

Table 14 The average activity concentration of radionuclides elements at the depth of 25 cm soil profile in study areas, Uppsala and Jämtland Counties.

Study area	Statistical parameters	K-40	Cs-137	Pb-210	Ra 226	Th-232	U-238
Vikdrolet	Average	833.60	6.57	562.67	411.27	34.93	533.73
	Range	695-950	0-17.7	315-897	227-681	28.1-39.1	355-765
	Std.	73.13	5.13	182.97	175.67	3.63	162.35
Myrviken	Average	675.80	9.47	258.07	250.53	28.79	378.71
	Range	564-883	2.56-14.6	180-341	209-291	18.6-42.2	82.6-595
	Std.	77.09	3.18	46.24	24.51	4.76	139.21
Backfors	Average	844.67	5.29	321.00	251.73	34.21	330.40
	Range	804-917	0-7.8	134-810	198-305	3.35-42.3	212-483
	Std.	36.62	2.45	207.56	29.81	9.06	76.86
Hallen	Average	791.00	6.45	80.97	22.93	42.66	365.67
	Range	692-932	1.32-11.8	52.3-129	11.9-45.8	32.4-101	212-602
	Std.	67.70	2.82	20.51	10.43	16.65	110.46
Lövstalöt	Average	997.20	63.25	84.42	61.13	60.44	485.73
	Range	867-1060	3.62-163	4.53-143	37.2-97.7	46.1-72.8	189-815
	Std.	53.57	52.89	35.70	16.22	6.62	204.62
Möjsjövik	Average	269.37	753.67	87.85	112.22	38.98	873.60
	Range	38.9-1770	386-1110	0-233	10.6-338	29.2-73.3	231-1830
	Std.	494.44	226.12	83.20	112.19	10.64	401.28

Table 15 The activity concentration of ^{137} Cs in KBq&m^2 at the depth of 25 cm soil profile in study areas, Uppsala and Jämtland Counties.

Study area	Bq/kg	Dry weight (kg)	Area (m^2)	KBq/m^2
Lövstalöt	63.25	0.04	0.0017	1.39
Möjsjövik	753.67	0.01	0.0017	5.03
Vikdrolet	6.57	0.06	0.0017	0.24
Myrviken	9.47	0.05	0.0017	0.29
Hallen	6.45	0.05	0.0017	0.18
Backfors	5.29	0.05	0.0017	0.15