Chapter 2

Review of Literature

This chapter describes the extensive studies and surveys carried out for measurement of natural and fallout radioactivity levels, concentration factors of different radionuclides in different matrices and background gamma radiation levels of different environs of the world. For the purpose, various journals and books available in the university library were referred. The articles published by various publishers, viz., Elsevier, Springer, Taylor Francis etc. were surfed through their websites. Documents, related to radioactivity, published by different international organizations, viz., Bhabha Atomic Research Centre (BARC), Atomic energy Regularity Board, India (AERB), World Health Organization (WHO), International Commission on Radiological Protection (ICRP), United States Environmental Protection Agency (USEPA), Agency for Toxic Substances and Disease Registry (ATSDR), European Commission (EC), United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR), International Atomic Energy Agency (IAEA) etc. were also consulted.

2.1 QUANTIFICATION OF RADIONUCLIDES IN WATER

Ability to sustain life and transport pollutants makes water an important parameter in environmental studies. Population of an area depends mainly on availability and quality of ground water for drinking and agricultural purposes. Radioactive materials may either be present naturally from geogenic sources or introduced anthropogenically from where they find their way into the water bodies. Ground sources may contain low levels of radioactivity as compared to surface waters which may contain fission products from fall-out, waste release and surface run-off etc. Radionuclides present in drinking water get incorporated into human food chain via ingestion and inhalation pathways. Estimates revealed that the total global average human exposure from natural sources is 2.4 mSv y⁻¹ (UNSCEAR, 1988, 2000). Ingestion pathways via food and water contribute about 0.3 mSv y⁻¹ to this average exposure. Total indicative dose is limited to 0.1 mSv y⁻¹ for adults through the ingestion of radionuclides via drinking water (WHO, 2011).

2.1.1 Quantification of Gross Alpha and Gross Beta Activity in Water

Gross alpha and gross beta activity estimation serves a preliminary step for the radiological characterization of drinking water to assess its suitability for human consumption. Alpha activity in natural water is imparted mainly due to uranium isotopes (²³⁴U, ²³⁵U and ²³⁸U) and ²²⁶Ra with small contribution of ²³²Th as its solubility is low (Osmond et al., 1992).
Similarly beta activity is due to short-lived daughters of $^{238}$U, $^{234}$Th and $^{234}$Pa up to a small extent but largely by $^{40}$K naturally and $^{90}$Sr anthropogenically (Blanchard et al., 1985). $^{226}$Ra as an alpha, $^{210}$Pb as beta emitter contributes significantly to the ingestion dose via consumption of water. The prescribed levels for gross alpha and gross beta activities are 0.5 Bq L$^{-1}$ and 1.0 Bq L$^{-1}$, respectively (WHO, 2011). If measured values of gross alpha and gross beta are below the specified levels in a water sample, then it is acceptable for its human consumption and no action is required to reduce its radioactivity. Accurate evaluation of the doses received by dietary intake is important from the view point of radiological protection. For strengthening the radiological screening of the drinking water World Health Organization (WHO, 2004) and US Environmental Protection Agency (USEPA, 2011) have issued several regulations on drinking waters.

Al-Amir et al. (2012) represented gross alpha and beta activities in tap water samples of Ammanaba, Jordan using liquid scintillation counter. The gross alpha activity ranged from 50 to 250 mBq L$^{-1}$ with mean $96 \pm 73$ mBq L$^{-1}$ and gross beta activity ranged from 188 to 327 mBq L$^{-1}$ with mean $251 \pm 53$ mBq L$^{-1}$ in Amman’s water. Much higher gross alpha and gross beta activities were recorded in Aqaba’s water. It was analysed that gross beta activities are generally higher than the corresponding gross alpha activities.

Bonotto (2011) applied combined gamma–alpha spectrometry technique for estimating gross alpha and beta radioactivities in water. The gross alpha activity ranged $1.1 – 41.9$ mBq L$^{-1}$ and the gross beta activity ranged $0.13 – 0.57$ Bq L$^{-1}$. Almost all gross alpha and gross beta activities were below the WHO (2011) guidance levels of 0.5 and 1.0 BqL$^{-1}$, respectively. In order to reduce the $^{40}$K contribution to the water radioactivity, no $^{40}$K activity subtraction from gross beta activity has been done.

Jobbagy et al. (2011) carried out the detailed study for measuring gross alpha and beta activity concentrations in spring waters in Balaton Upland, Hungary. A low background alpha/beta gas flow proportional counter was used to count the gross alpha and beta activities. The activity concentrations in the spring water for gross alpha ranged from 26 to 1749 mBq L$^{-1}$ and 33 to 2015 mBq L$^{-1}$ for gross beta. It was concluded that gross beta activity was higher than the alpha activity concentration. It was also concluded that 37.1% of the samples for gross alpha activity and 41.9% of the samples for beta activity concentration were above 100 mBq L$^{-1}$ and if consumed by human they can cause radiation risk. Gross alpha activity in natural water is mainly due to uranium and radium isotopes.

Gross alpha and gross beta activities of 30 different water samples collected from different water sources in Adana, Turkey were determined by Degerlier and Karahan (2010).
Activity concentrations of gross alpha in drinking water ranged 0.3 - 22.9 mBq L$^{-1}$ and gross beta activity ranged 18.9 - 290.7 mBq L$^{-1}$. Activity concentrations of sea water samples were measured from 800 to 3700 mBq L$^{-1}$ and from 700 to 6800 mBq L$^{-1}$ for gross alpha and gross beta, respectively. Combined alpha and beta radioactivity in seawater samples was high due to high salinity ratio of the Mediterranean sea water. Radioactive $^{40}$K isotope dissolved state causes high radioactivity in sea water thus gross beta activities in Mediterranean sea water samples were very high. Activity concentrations were 12.0 mBq L$^{-1}$ for gross alpha and 42.6 mBq L$^{-1}$ for gross beta in Seyhan Dam Lake and 5.0 mBq L$^{-1}$ for gross alpha and 245.3 mBq L$^{-1}$ for gross beta in Seyhan river.

Palomo et al. (2010) measured the gross alpha activity using a zinc sulphide (ZnS) scintillation counter and gross beta activity with a low background alpha/beta counter in water samples of a potable water treatment plant located in the South of Catalonia (Spain) for six years (2002 - 2007) in screening process. The observed gross beta activities for incoming and outgoing water to the plant were in the range of 80 – 180 mBq L$^{-1}$ and 60 – 160 mBq L$^{-1}$, respectively. The gross alpha activities ranged 50 – 80 mBq L$^{-1}$ and 50 – 70 mBq L$^{-1}$ for incoming and outgoing water samples, respectively. No significant difference was recorded for activities of ingoing and outgoing waters. It was concluded that the solubility of beta-emitters also decreases with pH, so these isotopes remain into water more than alpha-emitters. The principal contribution to gross beta activity was due to $^{40}$K, which is highly soluble in water.

Gross alpha and gross beta radioactivity measurements in bottled drinking water in Spain were performed by Palomo et al. (2007a). After gross alpha activity, gross beta activity, gross beta without $^{40}$K contribution had been determined. $^{226}$Ra and $^{224}$Ra were also determined for samples with high gross alpha activity values. The gross alpha activity was measured using a zinc sulphide (ZnS) scintillation counter and gross beta activity with a low background alpha/beta counter. The gross alpha activity ranged from <0.03 Bq L$^{-1}$ to 0.86 Bq L$^{-1}$ and gross beta activity ranged from <0.04 Bq L$^{-1}$ to 2.28 Bq L$^{-1}$. The gross beta activity without $^{40}$K contribution ranged from <0.04 Bq L$^{-1}$ to 0.88 Bq L$^{-1}$. It was observed that higher activities correspond to higher conductivity values (high salinity) and lower activities correspond to lower salinity. This is because high salinity favours the solubility of the radionuclides in the water.

Desideri et al. (2007a) reported gross alpha and gross beta activity in bottled water of Italy. The alpha activity was measured by counting in a ZnS(Ag) detector and the beta activity was measured in the low background beta counter. Gross alpha activity in water samples
ranged <4.0 - 277.5 mBq L\(^{-1}\) and gross beta activity ranged <24.9 - 930.0 mBq L\(^{-1}\). In five samples gross alpha activity exceeded the recommended WHO guideline activity concentrations for drinking water, whereas, gross beta activity in all the samples was in prescribed limit. The data showed that 76.5% and 64.7% of the samples presented, respectively, an alpha activity concentration and a beta activity concentration lower than 50.0 mBq L\(^{-1}\). Various other studies undertaken on gross alpha and gross beta quantification in water are given in Table 2.1.

2.1.2 Quantification of Tritium (\(^3\)H) Concentration in Water

Tritium is a hydrogen radioisotope which has both artificial and natural origins. It is formed artificially, mainly in Pressurised Water Reactors producing electricity, and is a by-product of the fission of uranium, which is the source of tritium in the environment. The natural cosmic contribution is smaller than the artificial contribution. Radiological impacts of tritium are considered due to its relatively long half-life (12.3 years) and easy uptake by humans. It has biological importance because of an isotope of hydrogen (Rohwer and Wilcox, 1976). Due to possible health hazard associated with higher concentration levels of tritium, investigations of tritium are required to calculate the risk (Straume, 1993; Okada and Momoshima, 1993; Gulden and Raskob, 2005). As tritium emits low energy β-particle which has a maximum range of only 6µm in water or tissue, prevents it from being external exposure risk. Moreover, it has a biological half-life of 10 days only, which also reduces its risk. Physical and chemical properties of tritium are almost similar to that of hydrogen. But tritium when bound to water becomes an essential part of photosynthesis and absorption reactions in plants and possibly uptake by humans (Elwood, 1971). The prescribed limit of tritium in drinking water is 6000 Bq L\(^{-1}\). Tritium levels in various water sources from different countries are given in Table 2.2.

2.1.3 Quantification of Uranium Concentration in Water

Radioactive elements are naturally present in different environmental matrices including the earth crust. Uranium naturally occurs in three isotopic forms in soil, water, plants, animals and human beings. \(^{238}\)U and \(^{235}\)U are the parent nuclides of two independent decay series, while \(^{234}\)U is a decay product of the \(^{238}\)U series. Despite low abundance, \(^{234}\)U and \(^{235}\)U isotopes are important due to their higher specific activity (Hakonson-Hayes et al., 2002). Uranium is used as fuel in nuclear energy production so it is considered as the most important radio-element in present era. Concentration of uranium in the earth crust has been reported to be more than that of some other elements, viz., Cd, Bi, Hg and I with an average
concentration of 4.0 µg g⁻¹ of rock (Katz et al., 1951). During the process of pedogenesis from igneous rocks, soils get enriched with uranium and from soil it gets transported to other environmental matrices such as water or air (USEPA, 2009). It has been reported that the average concentration of uranium in the ocean water is about 3.0 µg g⁻¹ (Rona et al., 1956). Uranium concentration in groundwater depends on several factors including lithological, geomorphological and other geological conditions of the area (Sridhar-Babu et al., 2008). Some anthropogenic activities like mining, milling, and nuclear fuel processing add uranium to various environmental components. Agrochemicals like phosphate fertilizers also contribute to uranium contamination of groundwater.

Oral, dermal and nasal are the major routes of human and animals exposure to uranium but oral route being the most important and 15% uranium intake in human body is from food intake and 85% is from drinking water consumption (Cothern and Lappenbusch 1983). Therefore, quantification of uranium in drinking water is important to assess its dose and risk to target population. Uranium has chemical and radiological effects and kidneys and lungs are two main target organs of uranium toxicity (ATSDR, 1990; ATSDR, 1999; WHO, 2008). A temporary damage to the kidneys may occur by consumption of about 0.1 mg kg⁻¹ of body weight of soluble natural uranium (Tanner, 1980). But chronic exposure to uranium via water ingestion even in low concentration can permanently damage the kidney (Zamora, 1998). ²²⁶Ra being a part of ²³⁸U decay series, is present naturally in 1:3×10⁻⁷ by weight. Its enrichment occurs in bones because it shares same group of periodic table as calcium, hence because long exposure of body tissues to radiation. Uranium emits alpha particles having high ionization energy which pose no risk in handling but can be hazardous if uranium is inhaled or ingested in significant amounts.

The uranium concentrations in various environmental matrices including soils, rocks, plants, water etc. have been reported in the past (Dyck, 1979; Dunn, 1981). It has been reported that uranium is higher than permissible limits in groundwater in certain parts of Northern India. The studies on natural uranium in water samples from various parts of India and world are given in Table 2.3 and 2.4, respectively.

2.1.4 Quantification of ¹³⁷Cs and ⁹⁰Sr in Water

Significant amounts of ¹³⁷Cs were released into the environment during almost all nuclear weapon tests and some nuclear accidents, most notably the Chernobyl disaster and the Fukushima Daiichi disaster. For an example the mean contamination of ¹³⁷Cs in Germany following the Chernobyl disaster was 2.0 to 4.0 kBq m⁻². ¹³⁷Cs reacts with water producing
caesium hydroxide, a water-soluble compound. The biological behaviour of caesium is similar to that of potassium and rubidium. After entering the body, caesium gets more or less uniformly distributed throughout the body, with the highest concentrations in soft tissue (Delacroix et al., 2002).

Strontium is present in the atmosphere in the form of wet or dry aerosols. $^{90}\text{Sr}$ is formed in nuclear reactors or during the explosion of nuclear weapons. It is considered a waste product due to very limited applications. Considerable amounts of $^{90}\text{Sr}$ were released in environment during atmospheric nuclear weapons tests conducted in the 1950s and 1960s and spread worldwide. Surface and underground water contain varying amounts of strontium. Most of the strontium complexes are soluble in water, therefore, can enter in food chains. $^{90}\text{Sr}$ emits a beta particle with, no gamma radiation and have chemical properties like calcium, therefore, it may concentrate in the bones and teeth. Its presence in bones can cause bone cancer, cancer of nearby tissues, and leukemia (ATSDR, 2004).

Radioactivity studies were carried out by Povinec et al. (2005) for investigation of concentration of $^{90}\text{Sr}$, $^{137}\text{Cs}$ and ($^{239,240}\text{Pu}$ in surface water of different latitudinal belts of the Pacific and Indian Oceans, under an IAEA's Co-ordinated Research Project "Worldwide Marine (WOMARS)". The activity concentration of $^{90}\text{Sr}$, varied from 0.1 to 1.5 mBq L$^{-1}$; $^{137}\text{Cs}$ concentration ranged from 0.1 to 2.8 mBq L$^{-1}$ and ($^{239,240}\text{Pu}$ concentration 0.1 to 5.2 µBq L$^{-1}$ in surface waters of the Pacific and Indian Oceans in the year 2000. It had been concluded that the data may be used as the average levels so that any new contribution from nuclear facilities, nuclear weapon test sites, radioactive waste dumping sites and from possible nuclear accidents can be identified.

Harnandez et al. (2007) investigated that the levels of total suspended particles (TSP) in the atmosphere and gross alpha, gross beta as well as $^{90}\text{Sr}$ activities in both the atmosphere and drinking water have been increased after the tropical storm Delta on November, 28 - 29, 2005 at Canary Islands (Spain) and western shores of Morocco. $^{90}\text{Sr}$ measurements were carried out with a low background proportional counter. The average activity concentration of $^{90}\text{Sr}$ in water has been reported 7.7 mBq L$^{-1}$ for a period of six years (July, 2000 – April, 2006), while $^{90}\text{Sr}$ concentration was reported about 440% greater, i.e., 34 mBq L$^{-1}$ after the Delta storm.

Inomata et al. (2009) used "HAM database - a global version" for estimation of spatial and temporal changes in $^{137}\text{Cs}$ concentrations in the surface water of oceans for a period 50 years (1957 – 2005). Four types of temporal variation in activity concentration of $^{137}\text{Cs}$ in the surface water had been reported. (i) As a result of high fallout in the North Pacific
Ocean due to atmospheric nuclear weapons tests during 1950s, $^{137}$Cs concentration decreased during the five decades: the decrease rate was higher during 1960s and 1970s than that after the 1970s. After the 1990s the activity concentration of $^{137}$Cs was almost constant. Also the latitudinal variation in activity of $^{137}$Cs became small during the period. (ii) The activity concentration of $^{137}$Cs changed within a constant range in the 1970s and 1980s, in the equatorial Pacific and Indian Oceans, suggesting the horizontal transport of $^{137}$Cs from areas of high global fallout in the mid-latitudes of the North Pacific Ocean. (iii) The activity concentrations of $^{137}$Cs decreased exponentially over the five decades, in the eastern South Pacific and Atlantic Oceans. (iv) In the Arctic and North Atlantic Oceans, including marginal sea, $^{137}$Cs concentration was strongly controlled by discharge from nuclear reprocessing plants after the late 1970s. From 1970 to 2005 the deceptive half-residence times of $^{137}$Cs in the surface waters of the global ocean ranged from 4.5 to 36.8 years. In the equatorial region, this time was longer than northern hemisphere. There was no significant difference along the latitudes of the apparent half-residence times in the Pacific and Indian Oceans. These results suggest that $^{137}$Cs in the North Pacific Ocean is transported to the equatorial, South Pacific, and Indian Oceans by the oceanic circulation.

Oura and Ebihara (2012) estimated radioactivity levels of $^{137}$Cs using HPGe detector in the river water samples from three major rivers spreading in or surrounding Tokyo Metropolis, viz., Edogawa River, Arakawa River and Tamagawa River. The measurement is important because the samples were collected immediately after the Fukushima Nuclear Power Plant accident on 28th March to 7th April, 2011. Edogawa River water was collected at Kanamachi and $^{137}$Cs activity concentration ranged from 0.6 to 2.5 Bq L$^{-1}$. The measured activity of $^{137}$Cs of water samples from Arakawa River at Asaka, varied from 0.4 to 1.4 BqL$^{-1}$. Tamagawa River water was collected at Hamura and $^{137}$Cs activity concentration measured in the range 0.4 – 0.9 Bq L$^{-1}$. The surface water of river must be contaminated by atmospheric $^{137}$Cs, if released due to accident and the activity in water should be higher. But coincidently rain occured in the region on 21st March and pollutant released in atmosphere due to accident in the form of aerosol or gasses get settled on the Earth.

Lavrentyeva (2014) reported the $^{90}$Sr activity in groundwater near a waste disposal site in Russia. The activity of $^{40}$K, $^{90}$Sr and $^{137}$Cs was measured in a beta-spectrometer for assessing $^{90}$Sr concentration from beta-radiation of its daughter $^{90}$Y radionuclide in water. The activity of $^{90}$Sr in well water during the different season of a year (2010) ranged from 0.5±0.3 to 38.0±7.0 Bq L$^{-1}$. The high values of $^{90}$Sr were observed due to contamination of ground water from the leakage of radioactive material of the nearby disposal site.
2.2 QUANTIFICATION OF ANTHROPOGENIC AND NATURALLY OCCURRING RADIONUCLIDES IN SOIL

Animal kingdom on the earth gets exposed from raditions emitted from numerous radioactive sources that are present in all types of soil, rocks and sediments. Naturally occurring radioactive materials are present since the creation of earth and anthropogenic radionuclides occur in soil mostly due to different human activities.

2.2.1 Quantification of $^{137}$Cs and $^{90}$Sr

$^{137}$Cs and $^{90}$Sr, the fission products and artificial radionuclides, in the environment have been mostly resulting from global fallout through nuclear weapon tests. These global fallouts have dispersed worldwide for the past 60 years (Godoy et al., 1998). These are the most intensively studied waste products in terrestrial and marine environments because of their long half-lives and high toxicity (Sysoeva et al., 2005). Although most of the radionuclides were released in the northern hemisphere, they were also transported partially to the southern hemisphere. The level of $^{137}$Cs and $^{90}$Sr are expected to be higher at locations that were close to their origin. It is now widely accepted that for creating environmental quality threshold values more information on baseline levels than those currently available is required.

About 881 above ground and 878 underground nuclear weapon tests were performed in second half of 20th century, which are the main cause of fallout radioactivity. In addition nuclear disasters such as Kyshtym Nuclear disaster of Russia (1957), the Windscale fire of UK (1957), SL-1 Experimental Power Station accident of USA (1961), Saint- Laurent Plant nuclear disaster of France (1969), Three Mile Island Accident of USA (1979), Buenos Aires accident of Argentina (1983), Chernobyl nuclear accident of Ukraine (1986), Goiania Accident of Brazil (1987), Tokaimura nuclear accident of Japan (1999), Fukushima Daiichi nuclear disaster of Japan (2011) etc. have added significant radioactivity level to the environment.

The transfer characteristics of $^{137}$Cs and $^{90}$Sr may change with soil properties such as pH, soil texture and organic matter, fallout properties of radionuclides with climatic circumstances, terrestrial use and management practices (Kirchner and Baumgartner, 1992; Kagan and Kadatsky, 1996). It may be possible that factors which account for increased plant concentration of radioactive caesium have similar consequence with radioactive strontium (Gastberger et al., 2000). Strontium is either bound in oxide or organic complexes, or to exchange sites on clay or organic matter while caesium gets fixed in clay minerals, due to this strontium is more transportable in most of the soil forms. In soils with high organic
matter due to greater cation exchange capacity $^{90}$Sr may be attached strongly (Coughtrey and Thorne, 1983) and $^{137}$Cs may be relatively more transportable (Rosen et al., 1999). Moreover in field studies the main problem is seasonal variation in climatic and hydrological conditions influencing migration rates making it more difficult to determine the reasons of different radionuclide behaviour in different soils. An uncertainty of the physico-chemical structure of the soils and deposition can be fundamental in field condition studies (Forsberg et al., 2000).

$^{137}$Cs quantification has been done globally by various researchers due to their interest in studying the global fallout. The weapon tests released about $10^{18}$ Bq of $^{90}$Sr and nuclear accidents added $10^{16}$ Bq of $^{90}$Sr into the atmosphere (Stamoulis et al., 1999). $^{90}$Sr contamination after the Chernobyl accident was reported negligible compared to $^{137}$Cs, in most countries. Very less field studies are limited to low level studies. According to UNSCEAR (2000) the $^{90}$Sr/$^{137}$Cs global fallout ratio is 0.64. Most of the studies available in the literature are on low levels of $^{137}$Cs and $^{90}$Sr radioactivity. Very little data is available on higher values.

Forte et al. (2002) measured radionuclides content of soil from 0 to 40 cm depth in Italy. $^{137}$Cs values ranged from about 7 to 425 Bq kg$^{-1}$; $^{90}$Sr values ranged from 0.2 to 17.4 Bq kg$^{-1}$. $^{137}$Cs content was always higher than that of $^{90}$Sr and their ratio ranges from 5 (Hill area) to 41 (Plane area). In most cases $^{137}$Cs concentration showed a maximum between 5 and 10 cm. Caesium content of the deepest section (30-40 cm) was generally lower than 5%. Grass radical is usually contained in the first 5 cm of soil; therefore, caesium uptake by grass roots may reduce the amount of radionuclide in the first layer. Strontium distribution in the first 40 cm of soil is almost constant. It can, therefore, be presumed that a relevant amount of $^{90}$Sr was actually contained in soil layers deeper than 40 cm, as a consequence of strontium high mobility of strontium in soil (Riise, 1990).

Rajaram et al. (2010) reported the evaluation for 25 years (1983-2008) of $^{137}$Cs and $^{90}$Sr monitoring data around Madras Atomic Power Station (MAPS), Kalpakkam, India. $^{137}$Cs activity ranged from 1.5 to 6.1 Bq kg$^{-1}$ and $^{90}$Sr activity ranged from 0.5 to 9.8 Bq kg$^{-1}$ in sediment samples. The activity concentrations of these radionuclides were low in the fallout range. Finally it was concluded that these are not related to the nuclear power station operation.

Kamath et al. (1968) had prepared baseline report for India’s first nuclear power plant at Tarapur, Maharashtra. $^{137}$Cs in soil was reported to be ranged from 1.7 to 8.3 Bq kg$^{-1}$ and $^{90}$Sr ranged from 0.8 to 5.1 Bq kg$^{-1}$. The radioactivity reported in that report was on natural and fallout of nuclear weapon testing and nuclear accidents such as such as Kyshtym Nuclear
disaster of Russia (1957), the Windscale fire of UK (1957), SL-1 Experimental Power Station accident of USA (1961) etc. because at that time no nearby nuclear reactor was in operation.

Nikolava et al. (2000) studied $^{137}$Cs activity level in different soil fractions dividing it in three different parts of bulk fraction, rhizosphere fraction and soil-root interface and found 413 Bq kg$^{-1}$ $^{137}$Cs activity in bulk fraction, 361 Bq kg$^{-1}$ in rhizosphere fraction and 1387 Bq kg$^{-1}$ in soil – root interface fraction. $^{137}$Cs activity was reported to be associated with living materials such as rootlets, mychorrhizae and mycelia. Mykhaylo et al. (2003) reported that $^{137}$Cs activity in Ukraine soils was nine times lower at 6 - 10 cm depth than 0.4 cm soil depth. Steiner et al. (2002) reported the role of fungi in uptake of $^{137}$Cs in forest soil with variation in dependence of mycelia on different layers of soil and inter-relationship with biological and physicochemical processes. $^{90}$Sr activity was found to be in range of 5.31 to 69.78 Bq kg$^{-1}$ in Bug River Valley, Poland by Solecki and Chibowski (2001).

Wallova et al. (2011) reported their work of the activity concentrations of $^{90}$Sr and $^{137}$Cs in soil samples in Austria. All soil samples were taken from undisturbed pasture-land. The maximum values for $^{137}$Cs and $^{90}$Sr were measured in the uppermost 0-4 cm of the core. The highest values were found on the Saualpe/Mirmiger Alm with about 2000 Bq kg$^{-1}$ for $^{137}$Cs and 160 Bq kg$^{-1}$ for $^{90}$Sr. This may be attributed to rain on this mountain after the Chernobyl accident. On the other hand the lowest $^{90}$Sr (8 Bq kg$^{-1}$) and $^{137}$Cs (77 Bq kg$^{-1}$) values were measured in Kaltenegg and in Rettenegg the respective concentrations of the radionuclides were higher but $^{90}$Sr/$^{137}$Cs ratio (0.05) was very small, probably due to local rainfall and representing a higher contribution from Chernobyl. From Mariapfarr, the soil samples showed lower activity levels but higher $^{90}$Sr/$^{137}$Cs ratios. As the weather was dry after the Chernobyl accident, the portion of nuclides may be attributed to global fallout.

Gjelsvik and Steinnes (2013) reported distribution pattern of Chernobyl fallout in Norway along with observations of decline in $^{137}$Cs activity in the surface soil. In that study, surface soil samples were collected from 0 - 3 cm depth at 464 sites in 1995 and 463 sites in 2005 all along Norway. The distribution pattern was found to be similar to original distribution of 1986 at both times (1995 and 2005) but the decrease in activity concentrations was not similar. Reduction was more in coastal areas than far inlands. The authors provided the reasoning of replacement of Cs ions with marine cations such as Mg$^{2+}$ and Na$^{+}$ while differences in precipitation chemistry were considered to influence the uptake of $^{137}$Cs in terrestrial food chains.

Daraoui et al. (2012) reported $^{137}$Cs activity in soil samples of Bavaria, Germany which ranged from 228 to 1028 Bq kg$^{-1}$. Khan et al. (2010) reported that $^{137}$Cs activity in soil
samples of Jhangar Valley, Eastern salt range, Pakistan was in the range of 1.3 to 46.8 Bq kg⁻¹. Average amount of ¹³⁷Cs added to the surface soil after the Fukushima accident was estimated to be 7.8 ± 1.7 Bq kg⁻¹. The depth profile of ¹³⁷Cs specific activity was found in log-normal shape with a peak between 5 cm and 7.5 cm below the ground (Park et al., 2013).

Lee et al. (2013) reported specific activities of ¹³⁷Cs in surface soil samples that varied from <0.1 to 17 Bq kg⁻¹. Soil samples were collected from 12 locations of Jeju Island, Korea between 2006 and 2008 and exhibited higher activity concentrations at the surface layer and then gradual decrease with depth.

The radio-ecological state of the forest ecosystem in the vicinity of the Ignalina Power Plant, Lithuania prior to decommissioning was analyzed with specific emphasis on ¹³⁷Cs and ⁹⁰Sr activity concentrations. Measurements of ¹³⁷Cs in a soil layer to a depth of 30 cm, every 5 cm, showed that this radionuclide is mainly accumulated at 0 - 5 cm depth, therefore, top soil samples to a depth of 5 cm from 1996 to 2008 were collected. The mean activity concentrations were in the range of 9.0 - 67.0 Bq kg⁻¹ and 0.6 - 17.1 Bq kg⁻¹ for ¹³⁷Cs and ⁹⁰Sr, respectively, in soil samples (Luksine et al. 2013).

After the disaster of Fukushima Daiichi Nuclear Power Plant in Japan an estimated $3.5 \pm 0.7 \times 10^{15}$ Bq of ¹³⁷Cs is thought to have been released into the sea. After the disaster, distribution of local ¹³⁷Cs anomalies on the seafloor near the nuclear power plant was investigated by Thornton et al. (2013). The measured data were compared with data after the Chernobyl accident pollution of 1986 and suggestions for regular monitoring programmes for adequate assessment were provided.

James et al. (2011) reported ¹³⁷Cs and ⁹⁰Sr activity in soil samples for Kaiga, Karnataka. At the Kaiga site four PHWR reactors are under operation. ¹³⁷Cs activity level in soil varies from <0.9 to 28.5 Bq kg⁻¹ dry weight and ⁹⁰Sr activity level in soil ranged <1.1 - 2.2 Bq kg⁻¹. ¹³⁷Cs activity level in the soils of Kaiga was also reported in the range of 4.7 - 21.4 Bq kg⁻¹ by Joshi et al. (2001). Siddappa et al. (1994) reported that ¹³⁷Cs activity in soil at Kaiga region ranged 6.02 - 1.13 Bq kg⁻¹ and ⁹⁰Sr activity 1.28 - 0.49 Bq kg⁻¹. Narayana et al. (1995) showed that ¹³⁷Cs activity in soil samples from coastal Karnataka was in the range of 1.6 - 21.6 Bq kg⁻¹ and ⁹⁰Sr activity was in the range of <1.0 - 0.7 Bq kg⁻¹. These studies concluded that ⁹⁰Sr activity within coastal Karnataka is low and ¹³⁷Cs activity in soil samples from coastal Karnataka is high compared to other Indian sites. Joshi et al. (2001) concluded that fallout activities are re-circulated into the soil at Kaiga through the decay of tree parts in this predominantly forest region. The higher concentration of ¹³⁷Cs in soil samples of coastal Karnataka was also attributed to the greater deposition of ¹³⁷Cs on soil as the region is
marked by heavy rainfall during the monsoon season (Narayana et al., 1995).

In Fukushima prefecture, an environmental radiation survey had been carried out in 2005 before the nuclear accident. In that study, $^{90}\text{Sr}$ activity concentration ranged from 0.2 to 20.4 Bq kg$^{-1}$ while $^{137}\text{Cs}$ activity concentration ranged from 1.3 to 660 Bq kg$^{-1}$. $^{134}\text{Cs}$ was not detected. The contamination might have been originated from past atmospheric nuclear weapon tests. After the nuclear accident, a small increment of $^{90}\text{Sr}$ was observed in the exclusion zone, close to the damaged nuclear power plant where $^{90}\text{Sr}$ activity concentration ranged from 1.4 to 80.8 Bq kg$^{-1}$ with average value 17.5 Bq kg$^{-1}$. Over the exclusion zone significant increment was not observed as $^{90}\text{Sr}$ activity concentration varied from 1.6 to 20.6 Bq kg$^{-1}$. In case of $^{137}\text{Cs}$ isotope, the rise was significant in and out of the exclusion zone, $^{137}\text{Cs}$ activity concentration ranged from 16.7 to 99,700 Bq kg$^{-1}$ and $^{134}\text{Cs}$ activity concentration ranged from 14.7 to 90,100 Bq kg$^{-1}$. (NRA, Tokyo, Japan, 2012).

$^{137}\text{Cs}$ and $^{90}\text{Sr}$ concentrations in agricultural soil samples were reported by Singh et al., (2015) within 15 km radius of Narora Atomic Power Station (NAPS), Narora, India. $^{137}\text{Cs}$ activity ranged <0.56 - 2.06 Bq kg$^{-1}$ in different soil samples. $^{90}\text{Sr}$ content was <0.095 Bq kg$^{-1}$.

2.2.2 Quantification of Naturally Occurring Radionuclides

There are numerous sources of radioactivity in the environment which emit radiation of different energies to which life on the earth is exposed. Naturally occurring radioactive materials in Earth crust and cosmogenic radionuclides in the atmosphere are permanent sources of radiation exposure to human beings, plants and other forms of life. Cosmic rays interact with atomic nuclei in the atmosphere and produce radionuclides while NORMs were formed during formation of the earth (Radenkovic et al., 2009). These radionuclides being unstable, decaying continuously into daughter products and emitting ionizing radiation of different energies and intensities to reach the stable state but due to adequately longer half-lives they survived since their creation (Mohanty et al., 2004). The natural sources are responsible for about 98% of the total radiation dose to the population (UNSCEAR, 1993).

Soil is upper most layer of Earth’s crust and constitutes not only of organic and inorganic compounds but also of radionuclides. Soil is formed by weathering of rocks and deposition of eroded matter. Natural radioactivity of soil keeps on changing due to biological processes, chemical reactions and human activities. $^{232}\text{Th}$, $^{235}\text{U}$ and $^{238}\text{U}$ decay series and singly occurring $^{40}\text{K}$ are the major causes of natural radioactivity in soil. $^{232}\text{Th}$, $^{235}\text{U}$ and $^{238}\text{U}$ finally decay to $^{208}\text{Pb}$, $^{207}\text{Pb}$ and $^{206}\text{Pb}$ through 10, 11 and 14 transformations, respectively. The state of secular equilibrium is achieved naturally by the radionuclides in these three
series, in which the radioactivity of all contributing elements within each series are nearly equal. Also, there is no significant change in the activity of the parent radionuclide under condition of secular equilibrium during many half-lives of its daughter products. The world average of specific activities is 35, 30 and 400 Bq kg\(^{-1}\) for \(^{238}\)U, \(^{232}\)Th and \(^{40}\)K, respectively (UNSCEAR, 2000).

The artificial radionuclides resulting from fallout of nuclear plant accidents such as Chernobyl accident, regular discharge of radionuclides from nuclear power plants and open nuclear weapon testing may also be present in soil. These natural or artificial radionuclides enter into the food chain once available for uptake by plants from the environment resulting in radiation dose to creature on Earth (Kabir \textit{et al.}, 2009). Local geology, location, altitude, weather conditions, geochemistry and human economic and technological activities etc. of each region in the world affects the concentration of natural occurring radioactive materials and artificial radionuclides present in the rocks and soil, which may cause variation in dose rate (Radhakrishna \textit{et al.}, 1993). There are some areas in the world such as Kerala, India, where levels of monazites and zircons minerals in soil are very high and known as high background radiation areas (UNSCEAR, 1993).

External exposure from ionizing radiation by natural sources is inescapable aspect of life on the earth because human beings are uncovered from gamma ray emitters present in building materials, air, water, soils, food and extra-terrestrial rays, which are more significant at higher altitudes (Thabyneh and Jazzar, 2012). Above certain level, natural radioactivity may cause harmful radiological health risk. These radionuclides cause health risk superficially due to their gamma ray emissions and internally by emission of alpha and beta particles from their daughter products. The dose rate as well as the absorbed dose decides severity and nature of the biological and clinical symptoms (Ajayi, 2008). Hereditary and somatic effects may occur due to excess of radiation exposure. Effects on human body may appear in the form of tissue damage, cancer and many other abnormalities in the functioning and shape of the organs. Genetic effects are those, which get transferred to future generations and may be caused due to damage of sperm and ovum (Mehra \textit{et al.}, 2010). Leukemia, bone cancer, breast cancer, lung cancer, thyroid are some common types of diseases, which may occur due to radiation effects (Abdullahi \textit{et al.}, 2013). The outdoor exposure of natural terrestrial radiation to human beings originates predominantly from the upper 30 cm of the soil only (Chikasawa \textit{et al.}, 2001). Distribution of radionuclides in soil is not uniform and respective information is noteworthy for radiation protection and measurement. If information about the distribution of radioactive elements is available then many diseases and
sicknesses can be efficiently managed. The studies on naturally occurring radionuclides in soil from various parts of world and India are discussed here.

a) Worldwide Study of Naturally Occurring Radionuclides in Soil

Huang et al. (2015) estimated the activity concentration of naturally occurring radioactive materials, in beach sand samples collected from Xiamen Island, China. The activity of radionuclides $^{226}$Ra, $^{232}$Th and $^{40}$K in the sand samples were estimated using a 3×3 NaI(Tl) gamma ray spectrometric system. The activity concentrations of $^{40}$K ranged from 197.4 to 487.6 Bq kg$^{-1}$, 6.5 to 41.4 Bq kg$^{-1}$ for $^{232}$-Th and 7.9 to 25.7 Bq kg$^{-1}$ for $^{226}$Ra in the beach sand. The mean of activity concentrations of $^{40}$K, $^{232}$Th and $^{226}$Ra was 401.0, 11.3 and 14.1 Bq kg$^{-1}$, respectively. It was concluded that the level of public exposure is insignificant in Xiamen Island beaches. Thus the investigated area can be regarded as a harmless area with common natural background radiation.

Hamadneh et al. (2015) estimated the radioactivity in storm dust samples collected from the top of buildings in Aqraba town, Jordan after the storm on May, 2012. The activity concentrations of $^{40}$K in dust samples ranged from 478 to 638 Bq kg$^{-1}$ with an average 547 ± 56 Bq kg$^{-1}$ while, reported value in soil had an average of 172 ± 21 Bq kg$^{-1}$. The activity in dust sample was roughly three times greater than in soil samples. The activity concentrations of $^{238}$U in dust ranged from 36 to 76 Bq kg$^{-1}$. The average activity of $^{238}$U in dust samples (49.3 ± 14.0 Bq kg$^{-1}$) was about two times higher than measured in soil samples (21.4 ± 4.4 Bq kg$^{-1}$). The activity of $^{232}$Th in dust samples ranged from 24.1 to 38.7 Bq kg$^{-1}$ with an average of 30.0 ± 4.9 Bq kg$^{-1}$. The activity of $^{232}$Th in dust was comparable to those in soil samples (average 26.0 ± 3.7 Bq kg$^{-1}$).

To prepare a database of topsoil properties, the natural activity in agricultural soils in Lombardia, Italy was estimated by Guidotti et al. (2015). The activities of $^{40}$K, $^{232}$Th and $^{238}$U activities ranged 242 - 1434, 20 - 70 and 24 - 231 Bq kg$^{-1}$, respectively. The soils formed from the magmatic rocks showed a higher activity of $^{238}$U than that of $^{232}$Th. The $^{232}$Th activities become lower than $^{238}$U where soils are creating from basic sedimentary rocks. In the mountainous and plain regions, it was observed that, there is extensive spatial dispersion of activities. The activity concentrations of $^{40}$K were always higher than that of other two radionuclides.

The activity concentrations of $^{40}$K, $^{232}$Th and $^{238}$U in soil samples along the Chao Phraya river basin were determined by Santawamaitre et al. (2014). The activity concentration of $^{40}$K, $^{232}$Th and $^{238}$U of the soil samples varied in the range 178.4 - 810.7,
12.9 - 142.9 and 13.9 - 76.8 Bq kg$^{-1}$ with the average mean values of 309, 30 and 29 Bq kg$^{-1}$, respectively. It has been concluded that variation of the activity concentrations of $^{40}$K, $^{232}$Th and $^{238}$U in some soil samples in the studied area was affected by high utilization of fertilizers and a geological structure variation. The activity concentration of $^{40}$K was within the worldwide range but due to high activity concentration in some soil samples from river areas, the activity concentrations of $^{238}$U and $^{232}$Th were found to be higher. While the mean activity concentrations of $^{40}$K, $^{232}$Th and $^{238}$U were comparable to the worldwide mean activity.

Abnormal high radioactivity in the black sand of the Chituc marine sand bank, north of the city of Navodari, Romania has been estimated by Margineanu et al. (2014). In the granulometric analysis, it was observed that the distribution of thorium and uranium is similar and highest radioactivity concentration belongs to 50 - 100 µm fractions for thorium ($^{228}$Ac = 1220 Bq kg$^{-1}$) and uranium ($^{226}$Ra = 1046 Bq kg$^{-1}$). Uranium and thorium radioactivity could be credited to the zircon and monazite fractions in the sand of 50 - 100 µm fractions. The activity concentration of natural $^{40}$K was found highest in the <50 µm fraction, suggesting that potassium element, by some means is related to clay minerals or to the other mineralogical components present in this fraction. On the basis of calculation of gamma index due to level of natural radioactivity, it was concluded that the visitor’s access should be restricted to the area and the sand could not be used as building material. Some other worldwide studies related to quantification of natural radionuclides in soil are encapsulated in Table 2.5.

b) Nationwide Study of Naturally Occurring Radionuclides in Soil

Yadav et al. (2015) reported the activity level due to the presence of $^{226}$Ra, $^{232}$Th and $^{40}$K in soil samples of Purola area in Garhwal Himalaya region. The measured activity of $^{226}$Ra, $^{232}$Th and $^{40}$K in collected soil samples of Purola was ranged between 13 – 55 Bq kg$^{-1}$ with an average of 31 ± 2 Bq kg$^{-1}$, 13 - 101 Bq kg$^{-1}$ with an average 30 ± 3 Bq kg$^{-1}$ and 150 – 1310 Bq kg$^{-1}$ with an average 583 ± 30 Bq kg$^{-1}$, respectively. The radium equivalent activity in collected soil samples was found to vary from 47 to 221 Bq kg$^{-1}$ with an average of 115 Bq kg$^{-1}$. The total absorbed gamma dose rate in this area was found to vary from 22 to 93 nGy h$^{-1}$ with an average of 55 nGy h$^{-1}$.

Srilatha et al. (2015) estimated the activity concentration of primordial radionuclides in soil samples collected from Ramanagara and Tumkur districts of Karnataka, India using gamma spectrometry (HPGe detector). The activity concentrations of $^{226}$Ra, $^{232}$Th and $^{40}$K in Ramanagara soil ranged 14.4 – 45.3, 42.2 – 111.0 and 507 – 1049 Bq kg$^{-1}$, respectively. The
activity concentrations of $^{40}\text{K}$, $^{226}\text{Ra}$, and $^{232}\text{Th}$ in Tumkur soil ranged $14.6 - 50.5$, $47.3 - 116.1$ and $389 - 1564$ Bq kg$^{-1}$, respectively. The $^{40}\text{K}$ is the largest contributor to the total activity for all soil samples with approximately $85.5 - 90.5\%$ of the total gamma activity. It has been also reported that the rocks in this region are granite rocks, which have more of naturally occurring radionuclides than other rock types.

Sartandel et al. (2014) estimated the natural radioactivity level around the proposed Bhabha Atomic Research Centre site at Visakhapatnam, Andhra Pradesh. The mean activity concentration of $^{238}\text{U}$, $^{226}\text{Ra}$, $^{232}\text{Th}$ and $^{40}\text{K}$ in soil samples ranged from 29 to 49, 22 to 49, 123 to 311 and 225 to 872 Bq kg$^{-1}$, respectively. The equilibrium between $^{238}\text{U}$ and $^{226}\text{Ra}$ radioactivities was observed in soil samples. In the studied area, activity concentration of $^{238}\text{U}$ in soil samples was found equivalent to the Indian average value of 29 Bq kg$^{-1}$ (UNSCER, 2000) but the activity estimated for $^{40}\text{K}$ and $^{232}\text{Th}$ was higher than the Indian average of 400 and 64 Bq kg$^{-1}$, respectively. The cause of higher radioactivity may be granite rocks which are good hosts for uranium, thorium and potassium associated in the form of minerals like monazite, samarskite etc. and these rocks are predominant in the south India.

Ravisankar et al. (2014) estimated the naturally occurring radioactive materials in soil collected from Tiruvannamalai, Tamil Nadu, India. The activity concentration of $^{40}\text{K}$, $^{226}\text{Ra}$ and $^{232}\text{Th}$ ranged from 363 – 724, <2.2 – 10.0 and 16 – 34 Bq kg$^{-1}$, respectively. It was concluded that the soil does not pose any radiation hazard if used as building material.

To estimate the natural radioactivity in soil, the samples were collected from the region around a Thermal Power Plant (NTPC) at Dadri (U.P.), India by Mahur et al. (2013). High resolution gamma ray spectroscopy (HPGe) was used to determine the activity concentration of $^{226}\text{Ra}$, $^{232}\text{Th}$ and $^{40}\text{K}$ in soil samples. The activity concentrations of $^{40}\text{K}$, $^{226}\text{Ra}$ and $^{232}\text{Th}$ in the soil samples ranged from 195 – 505, 32 – 121 and 19 – 45 Bq kg$^{-1}$, respectively. The activity concentrations of $^{226}\text{Ra}$ in soil samples were higher than the world average i.e. 33 Bq kg$^{-1}$ (UNSCEAR, 2000). While, the activity concentration of $^{232}\text{Th}$ was lower than world average i.e. 45 Bq kg$^{-1}$, whereas, the activity concentration for $^{40}\text{K}$ was comparable with the world average.

$^{226}\text{Ra}$, $^{232}\text{Th}$ and $^{40}\text{K}$ analyses have been carried out in soil samples collected from some areas of Punjab, India using gamma-ray spectrometry (Sabharwal et al., 2012). The reported activities of $^{226}\text{Ra}$, $^{232}\text{Th}$ and $^{40}\text{K}$ in the soil ranged 23.17 - 57.87 Bq kg$^{-1}$, 59.03 - 160.40 Bq kg$^{-1}$ and 228.06 - 501.03 Bq kg$^{-1}$, respectively. The average outdoor terrestrial gamma air absorbed dose rate due to natural radioactivity in the area was 84.65 nGy h$^{-1}$.

Mehra and Singh (2011) reported radioactivity of $^{238}\text{U}$, $^{226}\text{Ra}$, $^{232}\text{Th}$ and $^{40}\text{K}$ in soil of
different geological origins in Northern India. The activity ranged 22.63 - 116.23, 19.47 - 96.08 30.2 - 136.12 and 189.47 - 508.05 Bq kg$^{-1}$ for $^{238}$U, $^{226}$Ra, $^{232}$Th and $^{40}$K, respectively. Average activity concentrations of $^{238}$U, $^{226}$Ra and $^{232}$Th were higher and activity concentration of $^{40}$K was lower as compared to the worldwide average concentration. Various other studies conducted for quantification of natural radioactivity in soil in different parts of India are given in Table 2.6.

2.3 Quantification of Radionuclides in Grains, Vegetables, Fodder and Grass

The studies related to the radionuclide uptake by plants are essential because elements may accumulate in the edible parts of the plants and food consumption pathway of exposure is more dominant as compared to nasal and dermal routes (Sharma et al., 2007). Quantification of natural and artificial radionuclides is very important as it is the integral part of environmental health surveillance programmes and environmental radiation protection. So, it is necessary to understand the radionuclide uptake by plants, as radionuclides may be transferred to human beings through food chain. As environmental radiation monitoring is very important for environmental health surveillance, it becomes necessary to determine the contamination of radioactive materials in soil and crop samples. Rice, wheat and corn are main cereal food crops for global population out of which wheat and rice constitute the most important cereals. Rice is a critical food used by humans with respect to intake of radionuclides (IAEA, 2010), because most of the rice production is centred in Asia and there is an increase in demand with the increasing population. Therefore, it is very important to study various radionuclides present in soil and cereals along with their transfer factors from soil to grains, so that the matrices can be developed for remediation or prevention purposes. Similarly, animal produced food items also contribute to human diet. Therefore, it becomes equally important to study the transfer factor of various radionuclides for fodder crops.

Vegetables are also staple food items, so radioactivity quantification in them is also desired. Generally, the radioactivity concentrations in fruits over a period of years show a global reduction in magnitude depending on the type of radionuclide and human interferences in soil-plant system. In case of vegetables and herb fruits, the pathway of $^{137}$Cs and $^{90}$Sr after release from the source is governed by the type and variety of crop, the stage of the plant development and season and climate at the time of deposition. In agricultural ecosystems, the
uptake of radionuclides and their translocation to fruits and vegetables is affected by human management including agricultural practices emphasized more on changing physiology and translocation of nutrients, for the purposes of early cropping and yield higher in quantity as well as quantity (Carini, 2001).

2.3.1 Quantification of $^{137}\text{Cs}$ and $^{90}\text{Sr}$

The literature contains a number of studies dealing with the uptake of natural radionuclides but such studies are limited for $^{137}\text{Cs}$ and $^{90}\text{Sr}$. Even after being a major staple crops in Asia the data of $^{137}\text{Cs}$ in respect to rice and wheat grains are limited and the scarcest for $^{90}\text{Sr}$.

Singh et al. (2015a) reported the $^{137}\text{Cs}$ activity concentration in wheat sample collected over a period of three years (2010 – 2012) from the vicinity of Narora Atomic Power Station, Narora, India. The activity concentration of $^{137}\text{Cs}$ activity in wheat samples ranged from <0.20 – 0.48 Bq kg$^{-1}$. $^{137}\text{Cs}$ activity in 63% samples was below detectable limit. The activity concentration of $^{137}\text{Cs}$ and $^{90}\text{Sr}$ in rice samples collected over a period of three years (2009 – 2011) from an area around Narora Atomic Power Station, Narora, India was also estimated by Singh et al. (2015b). In rice grain activity of $^{137}\text{Cs}$ ranged from <0.2 – 0.65 Bq kg$^{-1}$ and activity of $^{90}\text{Sr}$ was below detectable limit (0.065 Bq kg$^{-1}$). $^{137}\text{Cs}$ activity in 64% rice grain samples was below detectable limit and no specific temporal or spatial trend was observed in $^{137}\text{Cs}$ activity in rice grains.

Karunakara et al. (2013) estimated the activity concentration of $^{137}\text{Cs}$ in rice samples of Kaiga as rice is an essential component of the diet for a majority of the population in India. A high resolution n-type HPGe detector (Canberra Industries Inc., Meriden, USA) was used for the study. The activity of $^{137}\text{Cs}$ in rice grains, straw and root ranged from 0.8 – 2.2, <0.1 – 3.9 and 0.2 – 7.1 Bq kg$^{-1}$, respectively. The mean value of the activity concentration of $^{137}\text{Cs}$ was higher in the root in all the fields. Ross et al. (2013) reported $^{137}\text{Cs}$ activity in fruits and vegetables collected from the vicinity of Kudankulum Atomic Power Station, India. $^{137}\text{Cs}$ in both fruits and vegetables was found to be below detectable limits, i.e. <0.01 for fruits and <0.02 for vegetables.

James et al. (2011) estimated the activity concentrations of $^{137}\text{Cs}$ and $^{90}\text{Sr}$ in leaf samples of 10 plant species collected from the vicinity of Kaiga, Karnataka site where four PHWR reactors are under operation. The main plants from which the leaves were collected include Congress, Eakky, Tikte, Teak, Badam, Cashewnut, Champa, Mango, Banana and Jamun. $^{137}\text{Cs}$ activity in plant leaves ranged from <0.6 to 9.1 Bq kg$^{-1}$ and $^{90}\text{Sr}$ activity ranged
from <1.2 to 4.5 Bq kg\(^{-1}\). The details are given in Table 2.7.

Sarap et al. (2015) estimated the activity concentration of \(^{137}\)Cs and \(^{90}\)Sr in root and stem of winter wheat. Samples were collected from a property of the Faculty of Agriculture in Belgrade in May 2013. The activity concentrations of \(^{137}\)Cs in stem and root ranged from <0.2 – 0.7 and 5 – 11 Bq kg\(^{-1}\) in fresh weight, respectively. The activity concentration of \(^{90}\)Sr in wheat’s stem and root ranged from 0.4 – 1.1 and <2.6 – 7.2 Bq kg\(^{-1}\) in fresh weight, respectively. The higher concentration of \(^{90}\)Sr and \(^{137}\)Cs is obtained in winter wheat roots than that in stem.

Table 2.7: \(^{137}\)Cs and \(^{90}\)Sr in leaf samples from kaiga, Karnataka (James et al., 2011)

<table>
<thead>
<tr>
<th>Sr. No</th>
<th>Plant Name</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Chromolaena odorata. King and Robinson. (Congress)</td>
</tr>
<tr>
<td>2</td>
<td>Calotropis gigantea. Linn (Eakky)</td>
</tr>
<tr>
<td>3</td>
<td>Cassia tora. Linn (Tikte)</td>
</tr>
<tr>
<td>4</td>
<td>Tectona grandis. Lf (Teak)</td>
</tr>
<tr>
<td>5</td>
<td>Terminalia catappa. L (Badam)</td>
</tr>
<tr>
<td>6</td>
<td>Anacardium accidental. Linn (Cashewnut)</td>
</tr>
<tr>
<td>7</td>
<td>Michealia champaka (Champa)</td>
</tr>
<tr>
<td>8</td>
<td>Mangifera indica L (Mango)</td>
</tr>
<tr>
<td>9</td>
<td>Musa paradisiaca L (Banana)</td>
</tr>
<tr>
<td>10</td>
<td>Syzygium cumin. L (Jamun)</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th></th>
<th>No. of Samples</th>
<th>(^{137})Cs Activity (Bq kg(^{-1}), dry wt.)</th>
<th>(^{90})Sr Activity (Bq kg(^{-1}), dry wt.)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>23</td>
<td>&lt;0.5 – 9.1</td>
<td>&lt;1.2 – 3.3</td>
</tr>
<tr>
<td>2</td>
<td>11</td>
<td>0.7 – 5.8</td>
<td>1.3 – 4.2</td>
</tr>
<tr>
<td>3</td>
<td>5</td>
<td>0.9 – 7.7</td>
<td>2.2 – 4.5</td>
</tr>
<tr>
<td>4</td>
<td>14</td>
<td>&lt;0.5 – 6.6</td>
<td>&lt;0.6 – 3.7</td>
</tr>
<tr>
<td>5</td>
<td>7</td>
<td>0.5 – 2.0</td>
<td>&lt;0.8 – 1.5</td>
</tr>
<tr>
<td>6</td>
<td>10</td>
<td>0.4 – 1.5</td>
<td>&lt;0.4</td>
</tr>
<tr>
<td>7</td>
<td>10</td>
<td>0.9 – 3.5</td>
<td>&lt;0.7 – 2.2</td>
</tr>
<tr>
<td>8</td>
<td>2</td>
<td>0.7 – 1.2</td>
<td>1.6 – 2.2</td>
</tr>
<tr>
<td>9</td>
<td>12</td>
<td>0.7 – 3.8</td>
<td>&lt;1.2</td>
</tr>
<tr>
<td>10</td>
<td>12</td>
<td>0.2 – 2.3</td>
<td>&lt;0.2 – 1.9</td>
</tr>
</tbody>
</table>

Radiological surveys at different parts of Marshall Islands, Australia were undertaken to determine \(^{137}\)Cs activity concentrations in common food plants. Peters et al., (2013) collected the samples of coconut (Cocos nucifera) as most common food plant. The activity of \(^{137}\)Cs was found significantly different between different atolls of the region. The mean activity concentrations of \(^{137}\)Cs in coconut juice were 0.34 Bq g\(^{-1}\), 0.027 Bq g\(^{-1}\), 0.007 Bq g\(^{-1}\) and 0.002 Bq g\(^{-1}\) at four different atolls.

\(^{137}\)Cs and \(^{90}\)Sr activity levels in plants around Tianwan Nuclear Power Plant (NPP) were studied by Lu et al. (2006). \(^{90}\)Sr was determined using a low-level gas flow alpha-beta counter and \(^{137}\)Cs was estimated using high resolution HPGe detector. The mean activity concentration of \(^{137}\)Cs in Pine Needle, Tea, Grass, China cabbage, Wheat and Rice was 0.1 ± 0.03, 0.27 ± 0.05, 0.65 ± 0.19, 0.019 ± 0.01, 0.33 ± 0.021 and 0.009 ± 0.007 Bq kg\(^{-1}\),

33
respectively. The $^{90}\text{Sr}$ mean activity in Pine Needle, Tea, Grass, China cabbage, Wheat and Rice was $7.7 \pm 4.1$, $3.0 \pm 1.1$, $2.1 \pm 0.3$, $0.23 \pm 0.06$, $0.084 \pm 0.045$ and $0.024 \pm 0.008 \text{ Bq kg}^{-1}$, respectively.

The activity concentrations of $^{137}\text{Cs}$ and $^{90}\text{Sr}$ in potato tubers samples collected from North West coast of England were estimated by Green et al. (1997b). The activity of $^{137}\text{Cs}$ in whole and flash mass ranged $0.003 – 0.03$ and $0.003 – 0.02 \text{ Bq kg}^{-1}$, respectively. $^{90}\text{Sr}$ activity in whole and flash mass ranged $0.01 - 0.08$ and $0.008 – 0.06 \text{ Bq kg}^{-1}$, respectively. It was concluded that $^{137}\text{Cs}$ activity was uniformly distributed throughout the tuber while half of $^{90}\text{Sr}$ activity was concentrated in peel. Kliment and Bucina (1990) estimated $^{137}\text{Cs}$ activity in fruits and vegetables in Czechoslovakia. The mean activity concentration of $^{137}\text{Cs}$ in fruits was 3.5 Bq kg$^{-1}$ and in vegetables it was 6.3 Bq kg$^{-1}$.

2.3.2 Quantification of Naturally Occurring Radioactive Materials

The concentrations of naturally occurring radioactive materials in paddy samples collected from Penang, Malaysia has been estimated by Alsaffa et al. (2015). The activity concentrations of $^{226}\text{Ra}$ in root, straw, husk, and grain of rice ranged from 4.5 to 12.0, 2.2 to 8.5, 1.1 to 5.1, and 0.5 to 2.8 Bq kg$^{-1}$, respectively; the corresponding activity of $^{232}\text{Th}$ ranged from 4.2 to 9.2, 1.8 to 4.7, 0.8 to 2.1, and 0.5 to 1.6 Bq kg$^{-1}$; whereas the $^{40}\text{K}$ activity concentration ranged from 48.8 to 261.7, 211.1 to 1165.4, 97.4 to 419.0, and 43.5 to 108.5 Bq kg$^{-1}$, respectively. The pattern of accumulation of $^{226}\text{Ra}$ observed was in the order root > straw > husk > grain. The 47% of the total activity of $^{226}\text{Ra}$ was found in the roots and about 9% in the grain. Similarly, 57% distribution of activity concentration of $^{232}\text{Th}$ was found in the root and 8% was in the grain. The distribution pattern of activity concentrations of $^{232}\text{Th}$ in different parts of the rice plant was found same as $^{226}\text{Ra}$ pattern. The distribution of activity concentration of $^{40}\text{K}$ was found maximum in the straw (59%) and the minimum in the grains (7%). Its distribution follows the pattern straw > husk > root > grain.

Natural radioactivity in various types of rice samples collected from Nineveh province, Iraq local market was estimated by Najam et al. (2015) using a NaI detector. Seven branded rice samples originating from different countries were collected. The results are reproduced in Table 2.8. It was observed that, the activity concentration of $^{232}\text{Th}$ ranged from 13.7 (Kalrose sample) to 71.9 Bq kg$^{-1}$ (Amber sample) with a mean value of 39.1 Bq kg$^{-1}$, the activity concentration of $^{226}\text{Ra}$ ranged from 51.2 (Amber sample) to 109.3 Bq kg$^{-1}$ (Kalrose sample) with a mean value of 84.1 Bq kg$^{-1}$, and $^{40}\text{K}$ activity concentration ranged from 231.8 (Kalrose sample) to 691.7 Bq kg$^{-1}$ (Amber sample) with a mean value of 435.3 Bq kg$^{-1}$.
Table 2.8: $^{226}\text{Ra}$, $^{232}\text{Th}$ and $^{40}\text{K}$ radioactivity (Bq kg$^{-1}$) in rice samples

<table>
<thead>
<tr>
<th>Sr. No.</th>
<th>Sample Name</th>
<th>Origin</th>
<th>$^{226}\text{Ra}$</th>
<th>$^{232}\text{Th}$</th>
<th>$^{40}\text{K}$</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Amber</td>
<td>Iraq</td>
<td>51.15</td>
<td>71.97</td>
<td>691.71</td>
</tr>
<tr>
<td>2</td>
<td>Nawras</td>
<td>Turkey</td>
<td>80.82</td>
<td>43.31</td>
<td>483.22</td>
</tr>
<tr>
<td>3</td>
<td>Al-deek</td>
<td>Thailand</td>
<td>80.20</td>
<td>40.70</td>
<td>500.76</td>
</tr>
<tr>
<td>4</td>
<td>Mahmoud</td>
<td>India</td>
<td>86.75</td>
<td>53.73</td>
<td>331.24</td>
</tr>
<tr>
<td>5</td>
<td>Kalrose</td>
<td>America</td>
<td>109.26</td>
<td>13.67</td>
<td>231.87</td>
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<tr>
<td>6</td>
<td>Al-alah</td>
<td>Pakistan</td>
<td>73.45</td>
<td>32.56</td>
<td>502.71</td>
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<td>7</td>
<td>Abu-alnessr</td>
<td>Uruguay</td>
<td>107.21</td>
<td>17.85</td>
<td>305.91</td>
</tr>
</tbody>
</table>

Patra et al. (2014) estimated radionuclides activity concentration in various food (milk, fruit, fish, egg and water) matrices collected from vicinity of proposed nuclear power plant site at Vishakapatnam, India. The results are given in Table 2.9. In different food matrices varying activity concentration of $^{238}\text{U}$, $^{232}\text{Th}$, $^{226}\text{Ra}$ and $^{40}\text{K}$ ranged from 0.002 to 10.6, 0.002 to 2.8, 0.1 to 7.2 and 3.0 to 110.8 Bq kg$^{-1}$, respectively.

Table 2.9: Radioactivity (Bq kg$^{-1}$) of natural radionuclides in food matrices

<table>
<thead>
<tr>
<th>Sr. No.</th>
<th>Food Matrix</th>
<th>$^{238}\text{U}$</th>
<th>$^{226}\text{Ra}$</th>
<th>$^{232}\text{Th}$</th>
<th>$^{40}\text{K}$</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Milk</td>
<td>1.0 - 5.9</td>
<td>1.1 - 2.7</td>
<td>0.3 - 2.8</td>
<td>3.0 - 26.1</td>
</tr>
<tr>
<td>2</td>
<td>Egg</td>
<td>0.05 - 6.60</td>
<td>0.1 - 3.3</td>
<td>0.1 - 0.6</td>
<td>14.9 - 33.9</td>
</tr>
<tr>
<td>3</td>
<td>Fruit</td>
<td>1.5 - 10.6</td>
<td>3.6 - 7.2</td>
<td>0.1 - 2.0</td>
<td>3.0 - 110.8</td>
</tr>
<tr>
<td>4</td>
<td>Fish</td>
<td>0.002 - 0.660</td>
<td>0.2 - 1.4</td>
<td>0.002 - 1.050</td>
<td>13.4 - 41.3</td>
</tr>
</tbody>
</table>

The concentration level of natural radioactivity in vegetables grown in high background radiation area of south-eastern part of Bangladesh was measured by Islam et al. (2014). In papaya samples the activity concentrations of $^{226}\text{Ra}$, $^{238}\text{U}$, $^{232}\text{Th}$ and $^{40}\text{K}$ ranged from 41.8 - 120.0, 18.6 – 111.0, 39.6 - 127.5 and 1030 - 2352 Bq kg$^{-1}$, respectively. The average activity concentration of $^{226}\text{Ra}$, $^{238}\text{U}$, $^{232}\text{Th}$ and $^{40}\text{K}$ was 80.9 ± 13.6, 64.8 ± 38.5, 83.5 ± 20.5 and 1691 ± 245 Bq kg$^{-1}$, respectively. The reported high concentration of $^{40}\text{K}$ was attributed to higher concentration of potassium in soil due to regression of seawater. Due to higher value of $^{40}\text{K}$ in papaya fruit, the dose received due to its consumption was four times higher than the world’s average value of fruits.
Natural radioactivity in root vegetables (Tapioca and Sweet Potato) grown in Selangor and Perak states of Malaysia was estimated by Asaduzzaman et al. (2014). The activity concentration of $^{226}$Ra, $^{232}$Th and $^{40}$K ranged between 72 - 141, 9.7 - 53.4 and 49.7 - 934 Bq kg$^{-1}$, respectively in Tapioca and 49.6 - 91.6, 5.3 - 23.4 and 259 - 3127 Bq kg$^{-1}$, respectively in Sweet Potato. It has been concluded that the activity was higher in samples collected from Puchong which is a mining area. The observed activity concentration of $^{226}$Ra and $^{232}$Th was high in Tapioca as compare to Sweet Potato while $^{40}$K activity concentration was comparatively much higher in Sweet Potato.

The activity concentration of $^{238}$U, $^{232}$Th and $^{40}$K measured using NaI(Tl) gamma spectrometer in most available wheat flour collected from the local markets in Iraq by Abojassim et al. (2014). It was observed that the specific activity in wheat flour samples was ranged from 1.08 to 12.53 Bq kg$^{-1}$ with an average 6.60 Bq kg$^{-1}$ for $^{238}$U, 0.12 to 4.30 Bq kg$^{-1}$ with an average 1.95 Bq kg$^{-1}$ for $^{232}$Th and 41.84 to 264.73 Bq kg$^{-1}$ with an average 133.10 Bq kg$^{-1}$ for $^{40}$K. It was concluded that consumption of wheat flour as the foodstuff is safe for people in Iraq.

Ross et al. (2013) estimated the activity concentrations of $^{40}$K, $^{228}$Ra and $^{226}$Ra in pulses and cereals samples cultivated around Nuclear Power Plant Kudankulam, India. The average $^{40}$K, $^{228}$Ra and $^{226}$Ra activities in pulses were 294.3, 1.07, 0.15 Bq kg$^{-1}$; while in cereals its corresponding values were 70.78, 0.47 and 0.09 Bq kg$^{-1}$, respectively. Changizi et al. (2013) reported $^{226}$Ra, $^{232}$Th and $^{40}$K activity in wheat and corn grains collected from Ilam Province, Iran. Mean values of $^{226}$Ra, $^{232}$Th and $^{40}$K activity were 1.67, 0.5 and 91.7 Bq kg$^{-1}$, respectively in wheat grains while 0.81, 0.85 and 101.5 Bq kg$^{-1}$, respectively in corn grains.

Alrefae and Nageswaran (2013) estimated the activity concentration of natural radionuclides using high resolution p-type HPGe gamma spectrometer, in Rice samples collected from the Kuwaiti local market. $^{238}$U was detected in 9 out of the 21 samples and ranged from 0.41 Bq kg$^{-1}$ (Coop, India) to 0.91 (Sun White, India) with an all-brand average of 0.62 ± 0.19 Bq kg$^{-1}$. $^{232}$Th was detected in all samples and ranged from 0.32 Bq kg$^{-1}$ (Lataste, France) to 0.62 (Dr. Moosa, India) with an all-brand average of 0.48 ± 0.10 Bq kg$^{-1}$. $^{40}$K was also detected in all samples and ranged from 32.9 Bq kg$^{-1}$ (Alquaem, Pakistan) to 101.0 (Rapunzel, Germany) with an all-brand average of 48.60 ± 18.34 Bq kg$^{-1}$. It was concluded that rice consumption available in Kuwait market is radiologically safe for the presence of the investigated radionuclides.

The concentrations of $^{238}$U and $^{232}$Th have been estimated in essential vegetables collected from Cameron Highlands and Penang, Malaysia by Aswood et al. (2013) using
neutron activation analysis technique. The samples of Tomato, Okra, Lattice, Pumpkin, Cucumber, Eggplant, Onion and Chilli were collected for the study. The $^{238}\text{U}$ activity concentration in Tomato, Eggplant, Lettuce, Pumpkin and Cucumber collected from Cameron Highlands was found to be $1.30 \pm 0.14, 1.90 \pm 1.09, 4.35 \pm 1.14, 1.90 \pm 1.09$ and $6.25 \pm 1.58$ Bq kg$^{-1}$, respectively and for $^{232}\text{Th}$ activity the corresponding values were $0.41 \pm 0.12, 0.41 \pm 0.12, 2.50 \pm 1.55, 0.41 \pm 0.12$ and $0.41 \pm 0.12$ Bq kg$^{-1}$, respectively. The $^{238}\text{U}$ activity concentration in Eggplant, Onion, Okra, Chilli and Tomato collected from Penang was $3.10 \pm 1.10, 1.30 \pm 0.14, 2.50 \pm 1.26, <1.30 \pm 0.41$ and $<1.30 \pm 0.41$ Bq kg$^{-1}$, respectively and $^{232}\text{Th}$ activity was $\leq 0.41 \pm 0.12$ Bq kg$^{-1}$ for all the samples collected. It was concluded that $^{238}\text{U}$ activity in Cameron Highlands has the trend of order: Cucumber > Lettuce > Eggplant > Pumpkin > Tomato. On the other hand, the trend of $^{238}\text{U}$ activity in Penang has the order: Eggplant > Okra > Onion > Tomato > Chilli.

Lenka et al. (2013) reported the concentrations of $^{232}\text{Th}$, $^{238}\text{U}$, $^{226}\text{Ra}$ and $^{40}\text{K}$ in staple cereals (rice and wheat), pulses and drinking water consumed by the population residing around high background radiation area located in Chhatrapur, Odisha, India. $^{232}\text{Th}$, $^{238}\text{U}$, $^{226}\text{Ra}$ and $^{40}\text{K}$ activities in collected samples ranged between $0.3 - 2.0$, $0.3 - 32.0$, $0.4 - 28.2$ and $14.3 - 957$ Bq kg$^{-1}$, respectively. Karunakara et al. (2013) estimated the activity concentration of natural radionuclides by rice grown in experimental field conditions outside Kaiga Nuclear Power Plant. As rice is a critical staple food in Asian countries and produced under flooded conditions, thus uptake of radionuclides for rice is affected by soil conditions. The estimated activity concentration $^{226}\text{Ra}$ and $^{228}\text{Ra}$ in rice grains were $<0.2$ Bq kg$^{-1}$ while it is $151.8$ Bq kg$^{-1}$ for $^{40}\text{K}$ and $4.0$ Bq kg$^{-1}$ for $^{210}\text{Pb}$.

Naturally occurring radionuclides in some food crops from high background radiation area on the Jos–Plateau, Nigeria were estimated by Jwanbot et al. (2012). A total of twenty food items including fruits and vegetables commonly used by residing population were collected across farmlands and vegetable gardens from the study area. The activity concentration of $^{40}\text{K}$ ranged from $12.36$ (Irish potatoes) to $56.92$ Bq kg$^{-1}$ (Spinach). The $^{226}\text{Ra}$ activity in different samples ranged from $1.46$ (Karkashe) to $10.42$ Bq kg$^{-1}$ (Water Melon). The activity concentration of $^{232}\text{Th}$ ranged from $1.53$ (Garden egg) to $6.85$ Bq kg$^{-1}$ (Vaat). The results show that all vegetables predominantly absorb $^{40}\text{K}$ more than $^{226}\text{Ra}$ and $^{228}\text{Th}$ radionuclides. As potassium being an essential element $^{40}\text{K}$ uptake was more favoured than other two radionuclides. Spinach absorbed the highest amount of $^{40}\text{K}$ concentration followed by cucumber and cabbage with values of $51.34$ and $42.37$ Bq kg$^{-1}$, respectively. Spinach and cucumber are leafy vegetables with broad leaves, having $^{40}\text{K}$ absorption points.
Saeed et al. (2012) estimated the natural radioactivity content in different varieties of rice (Faiza Basmati, White Glutinous, Black Glutinous, Siam, Kurnia and Utara) consumed by Malaysian population using HPGe detector. The overall activity concentration of $^{238}$U, $^{232}$Th and $^{40}$K in different types of rice varied as 5.1 - 18.3, 6 - 35.5 and 64.8 - 110 Bq kg$^{-1}$, respectively.

James et al. (2011) collected the Leaf samples of herbaceous plants and trees commonly seen in Kaiga, Karnataka environment in different seasons of the year and measured $^{226}$Ra and $^{40}$K activity concentrations in these leaves. The $^{226}$Ra activity in *Musa paradisiaca* L. (Banana), *Chromolaena odorata* (Congress grass), *Calotropis gigantea* Linn (Eakky) and *Cassia tora*. Linn (Tikte) leaves was in the range 0.5 - 4.78, <0.32 - 4.23, 0.58 - 4.32 and 1.39 - 5.94 Bq kg$^{-1}$, respectively and $^{40}$K activity ranged between 470 - 1202, <190 - 1085, 223 - 1656 and 165 - 769 Bq kg$^{-1}$, respectively.

Organically and conventionally grown winter wheat plants and their soil samples of corresponding fields’, collected during 2004-2007 from Belgium were monitored for the distribution of natural radionuclides using ultra low-level gamma-ray spectrometry (Lindahl et al., 2011). In root samples overall average activity concentrations were 30 ± 13, 1.3 ± 0.6, 1.4 ± 0.4 and 0.6 ± 0.2 Bq kg$^{-1}$ for $^{40}$K, $^{226}$Ra, $^{228}$Ra and $^{228}$Th, respectively. In stems $^{40}$K, $^{226}$Ra, $^{228}$Ra and $^{228}$Th overall average activity concentrations were 221± 84, 0.8 ± 0.3, 1.0 ± 0.3 and 0.20 ± 0.04 Bq kg$^{-1}$, respectively. Overall average activity concentrations in wheat grains obtained for $^{40}$K, $^{226}$Ra, $^{228}$Ra and $^{228}$Th were 115 ± 22, 0.10 ± 0.05, 0.15 ± 0.05 and 0.045 ± 0.026 Bq kg$^{-1}$, respectively. The observed soil-to-wheat concentration ratios for the different parts of the wheat plant in two agricultural systems revealed large variations in radionuclide activity concentrations between the sites and fields, but no significant difference between conventionally and organically grown wheat plants were observed.

Tufail et al. (2010) estimated the natural activity in wheat samples collected from Faisalabad in the Punjab province of Pakistan. The measured values of $^{40}$K activity concentration are higher than those of $^{226}$Ra and $^{232}$Th in the wheat grain samples. The activity concentration of $^{40}$K, $^{226}$Ra and $^{232}$Th was in the range of 95.7 – 146.9, 1.0 – 1.5 and 1.0 – 1.5 Bq kg$^{-1}$, respectively. It was concluded that the largest fraction among these radionuclides is of $^{40}$K and the smallest is of $^{232}$Th, the largest and the smallest transfer of $^{40}$K and $^{232}$Th, respectively into grain may be due to the smaller and larger densities of $^{40}$K and $^{232}$Th, respectively. The estimated dose due to consumption of wheat grain from the studied area was so small that from the radiation protection point of view its consumption can be treated as safe.
Concentration of natural radionuclides ($^{226}\text{Ra}$ and $^{40}\text{K}$) was estimated in food grains (rice and blackgram), vegetables and fruits in a high natural background area, Midalam, Kanyakumari, India (Shanthi et al., 2010). The radioactivity content of grains, vegetables and fruits in the study area ranged from 3.1 to 7.5, 0.03 to 1.2, 0.9 to 1.2 Bq kg$^{-1}$, respectively for $^{226}\text{Ra}$ and 120 to 482, 29.6 to 78.7, 30.4 to 136 Bq kg$^{-1}$, respectively for $^{40}\text{K}$.

Jibiri et al. (2007) analyzed the activity concentration of $^{40}\text{K}$, $^{238}\text{U}$ and $^{232}\text{Th}$ in different grain crops (Pearl millet, maize, rice, corn etc.) collected from a former tin mining location in a high background radiation area on the Jos-Plateau, Nigeria. Activity concentrations of $^{232}\text{Th}$, $^{40}\text{K}$ and $^{238}\text{U}$ were <2.0, 144 and 4.6 Bq kg$^{-1}$, respectively in millet; <2.0, 243.2 and 34 Bq kg$^{-1}$, respectively in maize; 7.6, 86, 5.2 Bq kg$^{-1}$, respectively in corn. All the studied radionuclides in Hungry rice were reported below detectable limits.

Mlwilo et al. (2007) estimated $^{40}\text{K}$, $^{232}\text{Th}$ and $^{238}\text{U}$ activities in staple food products (maize and rice) from various localities of Tanzania. Reported activity concentrations of $^{40}\text{K}$, $^{232}\text{Th}$ and $^{238}\text{U}$ in rice were 24.7, 3.8 and 5.0 Bq kg$^{-1}$ and in maize were 48.8, 4.1 and 13.2 Bq kg$^{-1}$, respectively. The activities of $^{40}\text{K}$ and $^{232}\text{Th}$ in widely consumed foodstuffs, rice and beans in Nigeria have been determined (Arogunjo et al., 2005). In this study activity concentrations of $^{40}\text{K}$ and $^{232}\text{Th}$ ranged from 42.29 to 74.46 and 4.0 to 10.5 Bq kg$^{-1}$, respectively in rice and from 257 to 298 and 3.5 to 6.0 Bq kg$^{-1}$, respectively in beans.

Pulhani et al. (2005) measured $^{226}\text{Ra}$, $^{238}\text{U}$, $^{232}\text{Th}$ and $^{40}\text{K}$ activities in wheat grain samples collected from two geographically different provinces (Punjab and Maharashtra) in India. $^{226}\text{Ra}$, $^{238}\text{U}$, $^{232}\text{Th}$ and $^{40}\text{K}$ activities in wheat grain from Punjab were in the range of 0.4 - 0.8, 0.016 - 0.06, 0.6 - 1.2 and 79.1 - 109.9 Bq kg$^{-1}$, respectively. $^{226}\text{Ra}$, $^{238}\text{U}$, $^{232}\text{Th}$ and $^{40}\text{K}$ activities in wheat grain from Mahabaleshwar were in the range of <0.2, 0.03 - 0.04, 0.1 - 0.2 and 116 - 130 Bq kg$^{-1}$, respectively.

### 2.4 Quantification of Radioactivity in Air

After Fukushima Daiichi accident, a series of car-born surveys were undertaken for the measurement of air dose rates in 100 km radius area around the Fukushima Dai-ichi Nuclear Power Plant, Japan (Andoh et al., 2015). An ionization-chamber-type survey meter was used in addition to the NaI(Tl) survey meter inside a 20 km radius from the Fukushima Dai-ichi Nuclear Power Plant where measured air dose rates were greater than 30 mSv h$^{-1}$. The first, second, third, fourth, and fifth surveys were conducted from June 4 to 13, 2011, from December 5 to 28, 2011, from March 13 to 30, 2012, from August 20 to October 12, 2012, and from November 5 to December 10, 2012, respectively. As concluded from the total
dose rate ratios, the air dose rates from the second to the fifth car-borne surveys decreased in comparison with those in the first survey by 34%, 41%, 53%, and 58%, respectively.

The natural activity concentrations in ambient particulate matter (PM2.5) aerosols and the associated inhalation effective dose to the public in Jeddah City, Saudi Arabia were estimated by Zytoon et al. (2014). The airborne PM2.5 concentration ranged from 16.5 - 188.4 µg m⁻³ with an average 50.81 ± 34.02 µg m⁻³. The activity concentrations of ⁴⁰K, ²³²-Th and ²³⁸U in air samples ranged from 11.0 to 19.4 µBq m⁻³ with an average 14.7 4 µBq m⁻³, 0.3 to 1.4 4 µBq m⁻³ with an average 0.7 4 µBq m⁻³ and 0.4 to 1.0 4 µBq m⁻³ with an average 0.6 4 µBq m⁻³, respectively. The annual effective radiation dose to the public due to inhalation of the airborne PM2.5 was in the range 15.03 - 58.87 nSv y⁻¹, depending on the age group. The dose rates were higher than the world average values in air reported in the UNSCEAR (2000) report.

Direct measurement of absorbed dose rate in air due to exposure from outdoor terrestrial gamma radiation was performed by Sharma et al. (2014) to establish a baseline data of annual effective dose and to assess the associated health risk from outdoor terrestrial gamma radiation along the rivers Alaknanda and Ganges of India. Absorbed dose rates in air ranged from 81.3 to 144 nSv h⁻¹ and annual effective dose ranged from 0.10 to 0.18 mSv y⁻¹ along these rivers. Excess lifetime cancer risk ranged from 0.375 × 10⁻³ to 0.662 × 10⁻³. Annual effective dose values were slightly higher than the world average of 0.07 mSv y⁻¹.

Carvalho et al. (2014) estimated the natural radioactivity in samples of smoke from vegetation fire collected from Viseu region, Centre North of Portugal, during late summer 2012. The average activity concentrations of ²³⁸U and ²³²-Th estimated in smoke free surface air aerosol samples collected from Viseu district, Portugal were 71.0 ± 3.7 Bq kg⁻¹ and 42.1 ± 2.7 Bq kg⁻¹, respectively. In smoke sample (collected in 2012) the activity concentrations of ²³⁸U and ²³²-Th ranged from 224 to 347 Bq kg⁻¹ and 95.7 to 203 Bq kg⁻¹, respectively. The estimated absorbed radiation dose to an adult exposed could be greater than 5 µSv d⁻¹ due to inhalation of smoke near forest fires for 24 hours and this value is more than 2000 times higher than the radiation dose from background radioactivity in surface air, and also higher than the radiation dose from ²¹⁰Po inhalation in a chronic cigarette smoker. Other than carcinogenic risk, other potential health effects of toxic substances present in smoke, and the health effect of fine particles inhalation may be very harmful at the measured concentrations of the smoke.

Radon concentration in the atmospheric air has been estimated by Shashikumar et al. (2009) around Mysore, Karnataka using Solid State Nuclear Track Detectors (SSNTD). The
higher concentration of radon in soil was observed near Chamundi Hills (5.94 kBq m\(^{-3}\)) and Karigatta (5.32 kBq m\(^{-3}\)) village. The concentration of radon gas in soil ranged from 0.60 (in summer) to 4.70 kBq m\(^{-3}\) (in monsoon). The ambient gamma radiation levels were measured using scintillometer and dose rate ranged from 0.009 to 0.018 mR h\(^{-1}\) with an average radiation level 0.012 mR h\(^{-1}\).

Mora et al. (2007) estimated the external natural radiations (cosmic and terrestrial) using NaI(Tl) scintillation counters along the roads in Costa Rica for the period July 2003–July 2005. The average effective dose was 46.88 ± 18.06 nSv h\(^{-1}\) for the total cosmic component and the average air-absorbed dose was 29.52 ± 14.46 nGy h\(^{-1}\) for the terrestrial component. The average total effective dose rate (cosmic plus terrestrial components) was 0.60 ± 0.18 mSv y\(^{-1}\). The effective dose rate per capita was found to be 83.97 nSv h\(^{-1}\) which is equivalent to an annual dose of 0.74 mSv and 2.29 mSv for Costa Rica population.

Kitto et al. (2006) represented the 22 years data of gross beta activity in surface air samples measured for the observation program as a means of assessing the release of artificial radionuclides (\(^{3}\)H and \(^{90}\)Sr) from the Nuclear Power Plant sites of New York State. Gross beta activity values ranged from 0.03 to 1.38 mBq m\(^{-3}\), with an arithmetic mean (AM) value 0.46 mBq m\(^{-3}\) and geometric mean (GM) value 0.44 mBq m\(^{-3}\). The similar values of AM and GM indicate a normal distribution of the gross beta activity in air samples. Results of samples collected during May and June, 1986 were excluded from these results because due to Chernobyl accident, gross beta activity was nearly 30 times greater than the normal values. It was concluded that the air borne radioactivity in New York was composed of naturally occurring radioactive materials only.

Akcay et al. (2006) estimated the gross alpha and beta activities in Ankara, Turkey airborne particulate samples in 2003–2004 collected from the top of the Ankara Nuclear Research and Training Centre. Air borne particulate samples were collected monthly during the years 2003 and 2004. WPC 9550 alpha and beta counting system were used for determining the radioactivity in air. The gross alpha activity ranged from 0.1 to 6.7 mBq m\(^{-3}\) with an average 2.02 mBq m\(^{-3}\) and gross beta activity ranged from 1.2 to 8.7 mBq m\(^{-3}\) with an average 2.85 mBq m\(^{-3}\).

2.5 Transfer Factor of Different Radionuclides from Soil to Vegetation

Transfer factor is measured as the ratio of concentration of radionuclides in plant or plant part to the concentration of that in soil. It is also depicted as bioaccumulation factor or concentration factor. It is a unit less quantity and can be used to measure the extent of
elements or radionuclides transferred to plant from the soil. It can be calculated for whole plant or root, shoot, leaves, fruits and seeds separately. The concept of transfer factor values came to the existence to relate the specific activity of plants to soil in case of emergency. Different types of data for different locations and plants have been generated in the last three decades, especially after the Chernobyl accident (26 April 1986 at the Chernobyl Nuclear Power Plant) (Wyttenbach et al., 1995). Later, it was developed on this idea that concentration of many elements in plants was in some way related to their concentration in soil. Soil to plant transfer factors of radionuclides are commonly used to estimate their food chain transfers and thus in dose assessment studies. One of the key parameters in environmental assessment is, therefore, the soil-to-plant transfer of a contaminant to food and fodder crops.

The transfer factor values for large area are not acceptable because of increase in soil types and crop varieties introduce increasing uncertainties in results. Transfer factor values are satisfactory only when they are considered for a restricted area and small time period. Further, transfer factor reported in the literature showed that the concentration of a radionuclide in a soil is not the only factor influencing its uptake by a plant but various and numerous other ecological variables affect the radionuclides transfer from soil to plants such as nutrient status, exchangeable K-content, pH, moisture content, major ions present in the soil-plant system, the effects of rhizosphere processes, soil micro-organisms, the factors influencing transport to roots and long-term uptake rates by roots (Ehlken and Kirchner, 2002).

Global studies on radionuclide uptake data are less, causing the results of statistical exploration to be misleading and unpersuasive. This is due to the fact that available studies do not cover all the variables and possible conditions in a comprehensive way. The values of transfer factor can be represented without a range or statement of uncertainties in the limited availability of data (which can further depend on other tangible and non-tangible variables). Besides, it does not take foliar uptake, dust deposition and microclimatic pollutants into consideration.

Some basic reasons for the variability in transfer factors for plants have been reported by U.S. Nuclear Regulatory Commission (USNRC, 2003) Office of Nuclear Regulatory Research. The same are reproduced below:

i. The physico-chemical characteristics of the element or radionuclide, i.e. whether it is able to undergo chemical transformation reactions.
ii. Exposure pathway including soil-water, water-foliage, atmosphere-foliage, soil-water-plant.

iii. Biological variables regarding different plant species and varieties.

iv. The chemical form of the element (elemental or compound, organic or inorganic) and its potential for speciation in the plant (e.g. formation of oxides, carbonates, ions etc.).

2.5.1 Transfer Factor of $^{137}$Cs and $^{90}$Sr in Crops, Vegetables, Plants etc.

The transfer factors (TF) of $^{137}$Cs and $^{90}$Sr from soil to root and soil to stem of winter wheat collected from Belgrade were estimated by Sarap et al. (2015). The measured transfer factor values for $^{90}$Sr for soil to roots ranged from 4.78 to 11.20 while for soil to stem were from 0.57 to 1.80. The calculated transfer factor values of $^{137}$Cs for soil to roots ranged from 0.20 to 0.39, while for soil to stem were from 0.011 to 0.025. It was concluded that $^{90}$Sr and $^{137}$Cs accumulations were higher in the wheat roots than in the stem. A good correlation was obtained between $^{90}$Sr TF to wheat roots and $^{90}$Sr TF to wheat stem ($r = 0.854$) while significantly lower ($r = 0.037$) correlation coefficient between $^{137}$Cs TF to root and $^{137}$Cs TF to stem.

Singh et al. (2015a) reported the transfer factor of $^{137}$Cs in wheat samples collected over a period of three years (2010 – 2012) from the vicinity of Narora Atomic Power Station (NAPS), Narora, India. Transfer factor of $^{137}$Cs in wheat grain samples ranged from 0.12 to 0.46 in NAPS region. The minimum transfer factor (0.12) was observed at Ramghat-1 location in 2011 located within 5–10 km radius area of NAPS. The maximum TF (0.46) was observed at location of Husainpur village (in <5.0 km radius area of NAPS). It was reported that transfer factor has decreased with cation exchange capacity, available potassium content and pH of the soil.

The concentration factors of $^{137}$Cs and $^{90}$Sr in rice sample collected over a period of three years (2009 – 2011) from an area around Narora Atomic Power Station, Narora, India were also estimated by Singh et al. (2015b). $^{137}$Cs activity was detectable only in 31% of collected rice grain samples. The TF of $^{137}$Cs in rice grain samples ranged from 0.10 to 0.40. The minimum transfer factor was observed for Asadpur location (located within 5–10 km radius area of NAPS) while maximum transfer factor was observed at Gangagarh location (in <5.0 km radius area of NAPS). As $^{90}$Sr activity concentration was below detectable limit in all rice grain samples therefore, it was not possible to calculate the transfer factor of $^{90}$Sr in the collected samples. It was concluded that the transfer factor of $^{137}$Cs increased with pH of the soil in that study.
Jazzar and Thabayneh (2014) estimated the transfer factor of $^{137}$Cs along with natural radionuclides from soil to plants and grass collected from the western north of West Bank environment Palestine. Transfer factor value of $^{137}$Cs was found to be less than other radionuclides in all plant species. The concentration factor ranged from 0.01 to 0.77 with an average value of 0.27. It was concluded that mean values of transfer factor of $^{137}$Cs in plant parts were found equivalent to range (0.02-3.2) as given by IAEA (2010). Also the higher concentration factor of $^{137}$Cs in some samples must be due to the dry weight deposition of atmospheric fallout over a long period of time due to strong adsorption power of soil.

Karunakara et al. (2013a) estimated soil-to-grass and grass-to-cow milk transfer factors for $^{137}$Cs by employing an experimental grass field setup. The transfer factor for $^{137}$Cs in case of soil-to-grass was found as $1.1 \times 10^{-1}$ and for grass-to-milk as $1.9 \times 10^{-2}$. It was concluded that TF of radiocaesium was similar to TF of stable caesium.

Transfer factor of $^{137}$Cs in rice grain, grown in Kaiga region, India was estimated by Karunakara et al. (2013b). The transfer factor values ranged from $6.6 \times 10^{-2}$ to $3.4 \times 10^{-1}$ with the mean value of $2.1 \times 10^{-1}$. On comparing with transfer factors of $^{137}$Cs, $^{40}$K and $^{210}$Pb, it was found that transfer of $^{40}$K was higher for arial parts while $^{137}$Cs accumulation was more in root as compared to arial parts.

Endo et al. (2013) measured transfer factors for $^{137}$Cs in different parts of rice produce including chaff, rice bran, brown rice and polished rice samples collected from rice paddy fields in Minami-Soma City, Japan after the Fukushima Dai-ichi Nuclear Power Plant (FDNPP) accident in March 2011. Rice crops were planted in the following May and harvested at the end of September. The transfer factor of rice ears was relatively low at 0.019 - 0.026 while of chaff, rice bran, brown rice and polished rice was found to be 0.049 (single sample of chaff), 0.10 -0.16, 0.013 -0.017 and 0.005 - 0.013, respectively.

Transfer coefficients of $^{85}$Sr and $^{137}$Cs for brown rice were estimated by Choi et al. (2011) in flooded rice fields near Wolsung nuclear site, South Korea. A solution containing $^{85}$Sr and $^{137}$Cs was applied at two stages; once being mixed with the soil 27 days before transplanting and again to the surface water one day after transplanting. Transfer factor values of $^{85}$Sr and $^{137}$Cs ranged from $1.6\times10^{-2}$ to $2.5\times10^{-2}$ and $2.2\times10^{-2}$ to $1.5\times10^{-1}$, respectively. Rice straw showed several times higher transfer factors of $^{85}$Sr than $^{137}$Cs.

James et al. (2011) estimated the concentration factors of $^{137}$Cs and $^{90}$Sr in leaf samples of 10 plant species collected from the vicinity of Kaiga, Karnataka site where four PHWR reactors are under operation. The transfer factor of $^{137}$Cs in herbaceous plant leaves ranged 0.23 - 1.27 in *Musa paradisiaca L* (Banana), 0.20 - 3.03 in *Chromolaena odorata*
(Congress), 0.23 - 1.93 in *Calotropis gigantea Linn* (Eakky) and 0.30 - 2.57 in *Cassia tora. Linn* (Tikte) while transfer factor of $^{90}\text{Sr}$ in congress plant leaves ranged 1.0 to 2.8. In other plant leaves $^{90}\text{Sr}$ activity was below detectable limits. The transfer factor of $^{137}\text{Cs}$ in tree leaves ranged 0.12 - 0.47 in *Terminalia catappa L* (Badam), 0.09 - 0.36 in *Anacardium accidentale Linn* (Cashewnut), 0.05 - 0.55 in *Syzygium cumin L* (Jamun), 0.12 - 1.53 in *Tectona grandis. L.f* (Teak), 0.39 - 1.16 in *Michelia champaka* (Champa) and 0.17 - 0.29 in *Mangifera indica L* (Mango) while $^{90}\text{Sr}$ transfer factor in Jamun and Teak tree leaves ranged from 0.18 to 1.58 and 0.42 to 2.67, respectively. In other tree leaves $^{90}\text{Sr}$ activity was below detectable limits.

The transfer values reported from all parts of the world have been analysed by International Atomic Energy Agency and provided with the most representative mean values of $2.9 \times 10^{-2}$ for cereals (separately $3.3 \times 10^{-2}$ in case of Maize), $6.0 \times 10^{-2}$ for leafy vegetables, $2.1 \times 10^{-2}$ for non-leafy vegetables, $4.2 \times 10^{-2}$ for root crops like radish, carrots etc., $5.6 \times 10^{-2}$ for tubers, $2.5 \times 10^{-1}$ in case of pasture for $^{137}\text{Cs}$, mainly ranging in the magnitude of $10^{-2}$ to $10^{-1}$ for edible plant parts. For $^{90}\text{Sr}$, transfer factors are in the range of $10^{-2}$ to $10^{0}$ scale, of $1.1 \times 10^{-1}$ for cereals (separately $3.2 \times 10^{-1}$ in case of Maize), $7.6 \times 10^{-1}$ for leafy vegetables, $3.6 \times 10^{-1}$ for non-leafy vegetables, $7.2 \times 10^{-1}$ for root crops like radish, carrots etc., $1.6 \times 10^{-1}$ for tubers, 1.3 in case of pasture (IAEA, 2010).

Uchida *et al.* (2009) reviewed transfer factor data from 1987 to 2000. It was concluded that while fallout $^{137}\text{Cs}$ had higher TF than stable Cs over several decades, geometric means of TF values for $^{90}\text{Sr}$ and $^{137}\text{Cs}$ under equilibrium conditions were not strongly affected by direct deposition and the distributions of these elements in soil components were in steady state. For $^{90}\text{Sr}$, the GMs of TF values were $0.4 \times 10^{-2}$ for white rice and $1.4 \times 10^{-2}$ for brown rice, whereas those for $^{137}\text{Cs}$ were $1.6 \times 10^{-3}$ for white rice and $4.1 \times 10^{-3}$ for brown rice.

$^{137}\text{Cs}$ and $^{90}\text{Sr}$ transfer factors for different plants around Tianwan Nuclear Power Plant, China were estimated by Lu *et al.* (2006). The radionuclide concentrations in pine needle, tea, pasture and vegetables were comparatively high and the concentration in rice was the lowest, the pine needle and tea were considered as the indicator species for $^{90}\text{Sr}$ and $^{137}\text{Cs}$. The mean values of transfer factors of $^{90}\text{Sr}$ and $^{137}\text{Cs}$ were 0.022 and 0.031 for rice, 0.066 and 3.83 for China cabbage, 0.0088 and 0.089 for wheat, 0.037 and 0.56 for grass, respectively. The observed transfer factor of $^{90}\text{Sr}$ from soil to plant was about one order of magnitude higher than that of $^{137}\text{Cs}$ in different plant species.

Monira *et al.* (2005) estimated the uptake of $^{137}\text{Cs}$ by wheat plant using NaI(Tl) and
LSS detectors during a pot experiment. Soil samples were collected from five different areas of Bangladesh namely Noail Kalibari of Amirpur under Sonargaon thana, Ganakbari under Savar thana, Southkalia under Dhamrai thana, Bhagyakul of Sreenagar thana and Chonkhola of Cox’s Bazar Sadar thana at a depth of 0 – 15 cm. The transfer factors for different parts of the wheat plant ranged from 0.05 to 0.114 in wheat straw, 0.066 – 0.133 in roots and 0.011 – 0.043 in wheat grains. The activity and transfer factor of radioactive caesium in wheat plants were found to be greatly affected by soil properties, i.e. clay content, available potassium, organic matter, cation exchange capacity, pH, exchangeable ions, etc. It was concluded that cation exchange capacity and Ca in soils influenced positively, while clay minerals, available potassium and organic matter, negatively affected the $^{137}$Cs activity concentration in wheat plants.

The transfer coefficients of artificial radionuclides for rice at Vietnam were estimated by Binh et al. (2003). The transfer factors of $^{137}$Cs and $^{90}$Sr ranged from $9 \times 10^{-4}$ to 1.4 with an average 5.05$x10^{-2}$ and 5.1$x10^{-2}$ to 3.3$x10^{-1}$ with an average 1.13$x10^{-1}$, respectively. It was concluded that range of transfer factor for $^{137}$Cs is higher but overall transfer factor value for $^{90}$Sr was more than that of caesium. Wang et al. (1998) estimated the transfer coefficients of the coarse hull, fine hull, brown rice and polished rice in the range of 0.09 - 0.76, 0.09 - 1.06, 0.027 - 0.222 and 0.005 - 0.088 for $^{90}$Sr and 0.12 - 0.63, 0.19 - 1.58, 0.07 - 0.27, 0.030 - 0.188 for $^{137}$Cs, respectively.

2.5.2 Transfer Factor of Naturally Occurring Radioactive Materials

Terrestrial food chains transfer of artificial and naturally occurring radionuclides has been studied extensively in the last decades. The concentration factors of naturally occurring radioactive materials in paddy samples collected from Penang, Malaysia have been estimated by Alsaffar et al. (2015). The calculated transfer factors of $^{226}$Ra and $^{232}$Th ranged from 0.006 - 0.036 with an average value of 0.018 and 0.004 - 0.014 with an average value of 0.009, respectively. The overall concentration coefficient of $^{226}$Ra was greater than that of $^{232}$Th, in spite of higher average activity concentration of $^{232}$Th in the soil. These differences can be associated with the difference in solubility of elements with different oxidation states. The transfer factor of $^{40}$K from soil to grain samples ranged from 0.074 - 0.472 with an average value of 0.235. The results showed that radionuclide concentration in grains is not linearly related to soil concentrations. The transfer factors of $^{226}$Ra and $^{232}$Th in grains were positively correlated to organic matter and negatively correlated to pH, cation exchange capacity, electrical conductivity and clay content of the
soil. The $^{40}$K uptake was found positively correlated with organic matter content, pH and cation exchange capacity of the soil but no significant correlation exhibited by the electrical conductivity and clay content of soil with $^{40}$K uptake by grains.

Harb et al. (2014) estimated the transfer factors of naturally occurring radioactive materials in 35 different plant samples collected from Saluga and Ghazal Protectorate area in Aswan, Egypt using high purity Germanium detector (HPGe). The transfer factor values ranged from 0.19 to 0.73 for $^{226}$Ra, 0.09 to 0.88 for $^{232}$Th, 0.10 to 0.76 for $^{228}$Ra and 0.31 to 2.95 for $^{40}$K. The high value of transfer factor was obtained due to high concentration of organic matter in the forest soils. The high transfer rate of natural radionuclides in the soil-plant confirmed the presence of an internal cycling of natural radionuclides in the ecosystem.

Transfer factor of natural radionuclides in root vegetables (Tapioca and Sweet Potato) grown in Selangor and Perak states of Malaysia was estimated by Asaduzzaman et al. (2014). The concentration factor values of $^{226}$Ra, $^{232}$Th and $^{40}$K ranged between 0.66 - 1.40, 0.14 - 0.73 and 0.79 - 1.29, respectively in Tapioca and 0.50 - 1.05, 0.22 - 1.40 and 0.61 - 3.50, respectively in Sweet Potato. The geometric mean of soil-to-tapioca and soil-to-sweet potato transfer factors for $^{40}$K were found to be significantly higher than that for the other nuclides in all locations except the transfer factor of tapioca in Lembah Beringin. It was concluded that it may be due to essentiality of the potassium element for the plant.

Jazzar and Thabayneh (2014) estimated the transfer factor of natural radionuclides along with artificial radionuclides from soil to plants and grass collected from the western north of West Bank environment - Palestine. For soil to plant, the average transfer factor (TF) values were found to be 0.60, 0.50, 0.31, and 1.70 for $^{226}$Ra, $^{238}$U, $^{232}$Th and $^{40}$K, respectively. For soil to grass the TF values were found to be 1.26, 1.12, 1.15 and 1.20 for $^{226}$Ra, $^{238}$U, $^{232}$Th and $^{40}$K, respectively. The average values of transfer factor from soil to grass were found to be higher than from soil to plant.

The transfer factor of naturally occurring radioactive materials in plants and vegetables collected from Savar and Manikganj, Dhaka, Bangladesh was reported by Gaffar et al. (2014). The transfer factor of $^{226}$Ra, $^{228}$Ra and $^{40}$K in plants collected from Savar ranged from 0.082 to 0.926 with an average of 0.404, 0.153 to 0.563 with an average of 0.388 and 1.274 to 3.741 with an average of 2.212, respectively. The highest TF values of $^{226}$Ra, $^{228}$Ra and $^{40}$K were found in Colocasia esculenta, Amaranthus tricolor and Colocasia esculenta, respectively and lowest TF values of corresponding radionuclides were found in Momordica...
charantia, Momordica charantia and Ipomoea aquatic, respectively. In Manikganj, the transfer factor of $^{226}$Ra, $^{228}$Ra and $^{40}$K in different vegetable samples ranged from 0.087 (Benincasa hispida) to 0.455 (Corchorus capsulrais) with an average 0.247, 0.061 (Benincasa hispida) to 0.806 (Corchorus capsulrais) with an average value 0.42 and 0.738 (Corchorus capsulrais) to 1.949 (Benincasa hispida) with an average of 1.582, respectively. TF value of $^{40}$K in different plants of Savar was about seven times higher than those of other radionuclides while in Manikganj it was 2 - 10 times higher. Positive correlation coefficient was found for $^{40}$K concentration with sand content, available K and Ca contents of the soil of Savar.

Karunakara et al., (2013b) estimated soil to rice transfer factors for natural radionuclides ($^{226}$Ra, $^{228}$Ra, $^{40}$K and $^{210}$Pb) for rice grown in natural field conditions of west coast, India, irrigated with water from the cooling water discharge canal of the Kaiga Nuclear Power Plant. $^{226}$Ra and $^{228}$Ra were below detectable limits in rice. The mean values of soil to white rice transfer factors were $1.8 \times 10^{-1}$ and $4.2 \times 10^{-3}$ for $^{40}$K and $^{210}$Pb, respectively. Transfer factor of $^{40}$K was higher in aerial parts of the plant as compared to the root while opposite trend was observed for $^{210}$Pb with higher retention in roots. The radionuclide concentration in plant samples were not linearly related to its concentration in corresponding soil samples.

The concentration factor values of $^{238}$U and $^{232}$Th have been estimated in essential vegetables collected from Cameron Highlands and Penang, Malaysia by Aswood et al. (2013). The values of transfer factor of $^{238}$U obtained for Tomato, Okra, Lattuce, Pumpkin, Cucumber, Eggplant, Onion and Chilli were 0.006, 0.023, 0.021, 0.009, 0.031, 0.009, 0.012 and 0.012, respectively and the corresponding values of transfer factor of $^{232}$Th was 0.002, 0.003, 0.002, 0.013, 0.002, 0.002, 0.003, 0.003 and 0.003, respectively. It was concluded that TF of $^{238}$U was higher than TF of $^{232}$Th in all the collected samples.

Radioactivity concentration of natural radionuclides (U, Th and K) of paddy crops from Malaysian fields were determined (Saeed et al., 2012). Transfer factor of radionuclides in studied rural and urban sites varied from 0.11- 0.20 and 0.05 -0.09 for uranium, 0.11- 1.43 and 0.09 - 4.12 for potassium and 0.02 - 0.15 and 0.01 - 0.11 for thorium, respectively. The outcomes satisfied the standards of IAEA.

Velasco et al. (2012) investigated soil to fruit transfer factors of $^{137}$Cs and $^{40}$K by tropical (Citrus limon B.) plants in Brazil. Results of the study revealed higher incorporation of potassium than caesium, by citrus fruits and transfer factor ranged from 0.54 to 1.02 for
\(^{40}\text{K}\) and from 0.02 to 0.06 for \(^{137}\text{Cs}\). The values were highest at early phase of fruit growth and decreased subsequently with development and maturity thus showing an exponential growth with time.

In environmental impact assessment of radioactive contaminations, transfer factors are the most important parameters required for mathematical modelling. Soil to leaf (herbaceous plants and trees) transfer factor for natural and fallout radionuclides were determined for Kaiga region in Karnataka state, India (James et al., 2011). Activity concentration of studied radionuclides was higher in leaves of herbaceous plants compared to that of tree leaves. Soil to leaf transfer factor varied from 0.03 - 0.65 for \(^{226}\text{Ra}\) and 0.32 - 8.04 for \(^{40}\text{K}\).

Mobility of uranium and thorium in soil and their bioavailability to crops (\(\text{Triticum aestivum L.}\) and \(\text{Secale cereale L.}\)) and native grasses, their translocation factors and temporal variations were studied in both field trials and greenhouse pot experiments (Shtangeeva, 2010). Significantly higher concentrations of thorium and uranium in roots compared to aerial parts were observed in plants grown on radionuclide enriched soils. Among the radionuclides thorium was found less bio-available compared to uranium and depends significantly on soil type. Correlations between radionuclide concentrations in soil, roots and plants were found species specific.

Tufail et al. (2010) assessed the transfer factor of natural radionuclides from soil to wheat grains grown under field conditions at Faisalabad, Pakistan. Reported transfer factor values in the study were 0.12 ± 0.02 for \(^{40}\text{K}\), 0.02 ± 0.004 for \(^{226}\text{Ra}\) and 0.04 ± 0.007 for \(^{232}\text{Th}\). Conceicao ao et al. (2009) monitored soil-to-sugarcane transfer factors of natural radionuclides (\(^{226}\text{Ra}, \ 232\text{Th}\) and \(^{40}\text{K}\)) in Brazilian sugarcane fields fertilised with phosphate fertilizers. Transfer factors of radionuclides occurred were in the following order \(^{40}\text{K} > ^{226}\text{Ra} > ^{232}\text{Th}\). Soil to vegetables (lettuce, carrots, and beans) transfer factor in Brazilian agricultural fields conditions varied from \(10^{-2}\) to \(10^{-1}\) for \(^{228}\text{Ra}\) and from \(10^{-4}\) to \(10^{-2}\) for \(^{238}\text{U}\) (Lauria et al., 2009).

Soil to vegetables and fruits (grown above a superficial uranium deposit) transfer factors of natural radionuclides were assessed at Khan Al-Zabeeb, Jordan (Al-Kharouf et al., 2008). Transfer factors of uranium in green parts (leaves, stems and roots) of the test crops were about two orders of magnitude higher than the fruits. Al-Masri et al. (2008) investigated baseline transfer factor values of \(^{40}\text{K}, \ 238\text{U}, \ 210\text{Pb}\) and \(^{210}\text{Po}\) from soil to some agricultural crops including fruits, vegetables, legumes and cereals at Syria. Vegetable leaves exhibited
higher transfer factor values for $^{210}$Po, $^{210}$Pb and $^{238}$U as compared to fruits and cereals leaves and transfer factor (TF) for $^{238}$U was found highest (0.1) in chickpea straw. Transfer factors for $^{210}$Po varied from $2.8 \times 10^{-2}$ in fruits of eggplant to $2.0 \times 10^{0}$ in grains of barley.

Akhtar et al. (2007) reported that soil to grain transfer factor of $^{40}$K was highest and that of $^{226}$Ra was lowest in wheat grown at agricultural farms in two districts in Pakistan. Transfer factors of natural radionuclides in wheat plants grown under natural field conditions on two morphologically different soils, from India were studied (Pulhani et al., 2005). Soil to wheat grain transfer factors varied from $6.0 \times 10^{-3}$ to $2.4 \times 10^{-2}$ for $^{232}$Th, $4.0 \times 10^{-4}$ to $2.1 \times 10^{-3}$ for $^{238}$U, $9.0 \times 10^{-3}$ to $1.6 \times 10^{-2}$ for $^{226}$Ra and $0.14 - 3.1$ for $^{40}$K. Roots of the wheat plants concentrated about 54 -75% of total $^{238}$U, $^{232}$Th and $^{226}$Ra activity and only 1-2% was found distributed in grains. On the other hand 57% of $^{40}$K activity accumulated in the shoots and 16% in the grains. The intake of radionuclides by consumption of wheat grains from the fields studied contributes a small fraction to the total annual ingestion dose received by man due to naturally existing radioactivity in the environment.

Soil to plant transfer and translocation factors of natural radionuclides for different plant species grown in soil contaminated with uranium mine tailings in China were studied (Chen et al., 2005). Highest transfer factor values of $^{238}$U were observed for Lupinus albus ($3.7 \times 10^{-2}$), $^{226}$Ra for white clover Trifolium pratense ($3.4 \times 10^{-2}$) and $^{232}$Th for Lolium perenne ($2.1 \times 10^{-3}$). Vera Tome et al. (2003) have studied the transfer factors for natural uranium isotopes ($^{238}$U and $^{234}$U), thorium isotopes ($^{232}$Th, $^{230}$Th and $^{228}$Th and $^{226}$Ra in plant samples (grass-pasture) growing in granitic and alluvial soils around a disused uranium mine located in the Extremadura region, south-west Spain. Affected and non-affected areas of the mine presented large differences in activity concentrations of radionuclides of uranium series.

2.6 Conclusions

Various studies conducted in last 30 years related to quantification of naturally occurring radioactive materials and artificial radionuclides in various environmental matrices have been compiled in this Chapter. The data showed that radioactivity contribution in various environmental matrices and transfer factors of radionuclides from soil to crop are highly variable with time and space. Each study has its importance for that particular area in which it has been undertaken. The observation of one particular area may not be applicable for other area due to variation in climate and soil properties. This emphasizes the need to undertake such studies in restricted area and smaller time period.