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Assessment of Radiological Hazards and Trace Element Contents in Seagrasses and their Effects on DNA Damage in the Gulf of Aqaba, Jordan

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ABSTRACT: This work assesses the activity concentrations of gamma-emitting radionuclides and trace elements in two seagrass species collected from the Gulf of Aqaba, Jordan. The measurements were carried out using gamma spectroscopy, atomic absorption spectrometry (AAS), and the comet assay to evaluate DNA damage. The activity concentrations (mean ±) for ^{40}K, ^{226}Ra, ^{228}Ra, ^{238}U, and ^{210}Pb were 337.1±80.0 Bq/kg, 47.9±28.2 Bq/kg, 20.3±4.2 Bq/kg, 45.8±5.1 Bq/kg, and 93.4±17.3 Bq/kg, respectively. Most of the maximum concentrations were recorded at SG1, SG3, and SG5. All calculated radiological hazard indices, including Raeq, D_R, and H_{ex}, were below the recommended limits, except at one site (SG5), indicating negligible radiological risk. Analysis of heavy metal levels (Fe, Mn, Zn, Cu, Cr, Co, Cd, Ni, and Pb) revealed that Cu, Cr, Co, Cd, Ni, and Pb concentrations were below the detection limit of the AAS method. Zn and Mn concentrations were within permissible limits at all sites. Fe concentrations were also within permissible limits at selected sites; however, elevated Fe levels were observed at SG3 and SG4, indicating site-specific contamination. Results of the DNA damage assessment showed that samples collected from SG4 and SG5 had the longest comet tail lengths, indicating greater DNA damage than samples collected from the other sites. DNA damage was weakly related to the radionuclides contents and showed no correlation with heavy metal concentrations. This suggests that other factors may contribute to the observed DNA damage. These results highlight the need for continuous monitoring, site-specific risk assessment, and regulatory control to reduce contaminant exposure and protect marine ecosystems in the Gulf of Aqaba.

KEYWORDS: Marine Pollution; Seagrass; Natural Radionuclides; Heavy Metals; Genotoxicity.

1. INTRODUCTION

The marine environment plays a critical role in maintaining global ecological balance and supporting biodiversity. It provides essential ecosystem services, including oxygen production, climate regulation, and habitats for numerous species that support human livelihoods. Marine ecosystems, such as coral reefs, mangroves, and seagrass beds, serve as biodiversity hotspots and act as natural buffers against coastal erosion and extreme weather events. Oceans also contribute substantially to global food security by supplying an important share of dietary protein. Despite their importance, marine environments face increasing threats from human activities, including pollution, overfishing, as well as climate-change-related pressures such as ocean acidification and rising sea temperatures. Therefore, monitoring and protecting marine ecosystems are essential for maintaining their ecological functions and long-term sustainability.

Analyzing radiation levels and heavy metal concentrations in marine environments provides a reliable indicator of marine pollution (Abdallah et al. 2025). Contamination arises from various sources, including atmospheric deposition, direct discharges of polluted water from anthropogenic activities, and inputs from surface water systems. These pollutants can significantly impact marine ecosystems, necessitating their assessment to monitor and mitigate environmental degradation effectively.

The Gulf of Aqaba is Jordan's only marine outlet. It is notable for its rich and diverse coral reef systems. Various activities and industrial operations are distributed along the Aqaba coast, with particular emphasis on the phosphate port situated in the southern region of the coastline. Because of its ecological significance, this study addresses the hazards related to radiation and heavy metal contamination in two different types of seagrass growing in the Gulf of Aqaba and their impact on DNA through the evaluation of genotoxicity.

Various studies have been conducted to evaluate the radiological content in soil, water, food, and other environmental matrices (Okoor et al. 2019, Hamadneh et al. 2017, Abdallah et al. 2020, Abdallah et al. 2022). The importance of these studies is related to the fact that natural radiation exposure levels vary by approximately a factor of three worldwide (UNSCEAR 2000).

The activities of nuclear fuel cycle are considered the main source of radioactive residues. In addition, industrial processes, including mining, processing of industrial minerals such as phosphate and building materials, and production of non-nuclear fuels such as oil and gas, may generate radioactive residues (IAEA 2013).

The existence of radionuclides in the environment is related to natural and industrial sources. ^{40}K, ^{226}Ra, ^{234}Th, ^{210}Pb, and ^{238}U are examples of naturally occurring radionuclides. ^{137}Cs is one of the nuclides produced during nuclear tests or accidents such as Chernobyl and Fukushima.

The rapid development of industry has led to a significant increase in heavy metal pollution, including cadmium (Cd), copper (Cu), lead (Pb), nickel (Ni), and mercury (Hg), which contaminate soil and sediment. These heavy metals can enter biological systems through different exposure pathways because of their persistence and accumulation in the environment. While some heavy metals are essential for human health in small amounts, excessive exposure can be toxic and cause adverse health effects. Prolonged exposure to heavy metals has been linked to various health issues, including cancer. For instance, mercury (Hg) exposure can cause damage to the brain and liver, and pose risks to pregnant women through ingestion or inhalation. Similarly, long-term exposure to lead (Pb) can lead to cardiovascular diseases, while chromium (Cr) exposure has been associated with breast cancer-related deaths. Cadmium (Cd) exposure, on the other hand, can cause chronic kidney disease (Yuhu & Qinxian 2021).

Heavy metals in marine ecosystems pose significant threats to the health of marine biota and ecosystem balance. Seagrasses can accumulate fine sediments and particulate matter from natural and human-induced sources, which are often enriched with heavy metals. They can therefore act as biogeochemical barriers by trapping metals and reducing their further dispersal (El-Metwally et al. 2017, Harikumar et al. 2009, Dembitsky 2003, Jazzar & Thabaynch 2014, Khalafallah et al. 2019).

Exposure to radiation represents a significant threat to genomic integrity, as it can induce a wide spectrum of DNA lesions, including single-strand breaks, double-strand breaks (DSBs), base damage, and crosslinks. This occurs in nature through direct energy deposition in DNA molecules or indirectly through

production of free radicals by water radiolysis (Ravanat 2018). The accumulation of DNA lesions may be difficult to repair accurately and can lead not only to immediate mutagenesis but also to radiation-induced genomic instability that may persist across multiple cell generations (Suzuki et al. 2003).

This study highlights the presence of radiation and heavy metal contamination in seagrasses at different sites in the Gulf of Aqaba, Jordan and its association with DNA damage. The findings clarify the relationship between persistent pollutants and damage to genetic material, which may result in long-term biological and ecological impacts. The study provides scientific evidence linking environmental contamination to genetic damage and supports informed decision-making for environmental protection and management.

2. MATERIALS AND METHODS

2.1 Sample Collection

Seagrass samples were collected from five sites in the Gulf of Aqaba during the summer of 2023. The sites were selected to represent different ecological conditions and degrees of anthropogenic impact. Two seagrass species, *Halophila stipulacea* and *Halodule uninervis*, were identified and collected at Sites 1 and 2, while only a single species was present at each of the other sampling sites. For each sampling site, five independent seagrass samples of the same species were collected and analyzed, and the mean values were calculated and used to represent the overall results.

These sites are characterized by varying levels of human activity, including tourism and industrial operations, while one site (SG2) serves as a green area, with minimal apparent pollutant inputs and anthropogenic disturbance. Fig. 1 shows the location of these sites. The description of the collected seagrass samples is summarized in Table 1.

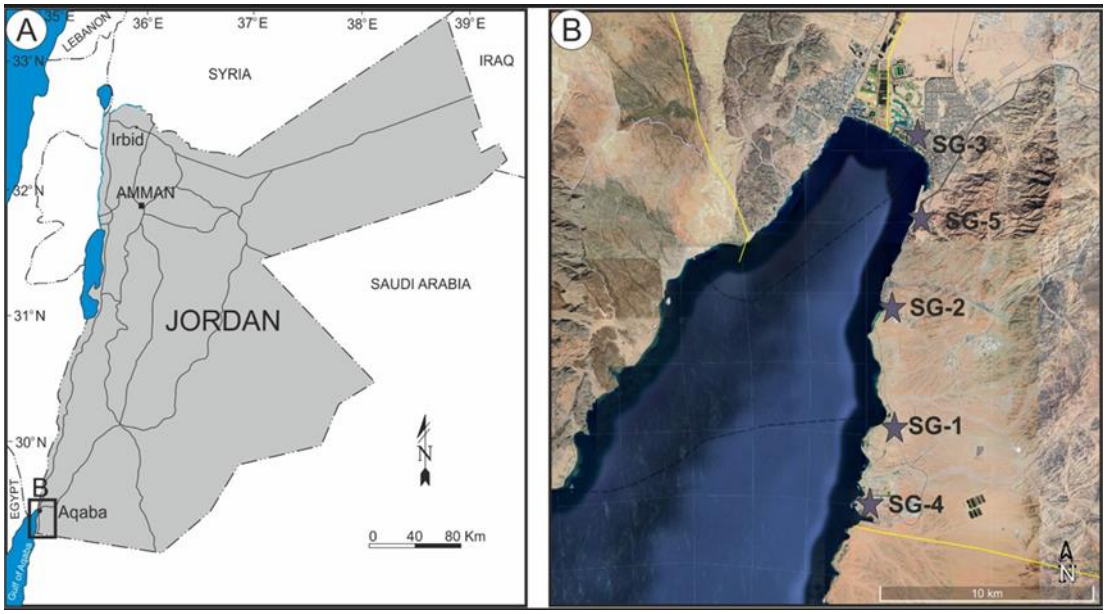

Fig. 1: Sampling Site Location

Table 1: Description of sampling site locations.

Sample	Depth (m)	Temperature (°C)	Visibility (m)	Location
SG1	5	28	28	N 29° 24' 33.56" E 34° 58' 39.29"
SG2	1.5	29	28	N 29° 27' 29.66" E 34° 58' 35.92"
SG3	5	28	28	N 29° 32' 11.83" E 34° 59' 19.03"
SG4	5	25	28	N 29° 22' 45.93" E 34° 58' 36.64"
SG5	5	28	28	N 29° 30' 42.35" E 34° 59' 50.88"

2.2 Sample Preparation and Spectroscopy Measurements

2.2.1 Radioactive analysis

Seagrass sampling and preparation procedures were performed according to established methodologies reported in the literature for marine radionuclide studies, consistent with the technical recommendations

provided by the IAEA for radionuclide analysis in marine biota (IAEA 1989, IAEA 2023). After sample collection, the samples were thoroughly washed to remove any adhering particles or contaminants. They were dried in a well-ventilated area, away from direct sunlight, for a sufficient time to ensure complete moisture removal. The dried samples were ground into a fine powder, transferred into 490 mL Marinelli beakers, and sealed carefully away from radiation sources for thirty days to ensure secular equilibrium between parent radionuclides and their corresponding daughter products.

The activity concentrations of the radionuclides were measured at the Jordan Atomic Energy Commission (JAEC) using gamma spectroscopy. A High Purity Germanium detector (HPGe) from Canberra, GX4020, USA was used to detect radionuclides. The efficiency of the detector was 40% at 1.33 MeV of ^{60}Co and resolution was 1.00 keV (FWHM) at the same energy peak. A standard source (CBSS 2, Canberra, USA) of multiple gamma emitters was used for calibration. The detector is shielded by a cylindrical lead layer to reduce the background radiation. A counting time of 24 h was employed for each sample to obtain reliable counting statistics, with the detector dead time kept below 10%. Background measurements were performed under identical experimental conditions using an empty counting container. The resulting background spectra were subsequently subtracted from the measured spectra of each sample to obtain the net activities of the radionuclides. Efficiency calibration was performed using a multi-nuclide standard source certified by the Czech Metrology Institute (Certificate No. 1035-SE-40701-25, Type: CBSS 2), containing an energy range from Am-241 (59.5 keV) to Zn-65 (1115.5 keV). The JAEC followed ISO 2009. The results of the detected radionuclides were interpreted using Genie-2000 software (Canberra, USA).

2.2.2 Heavy metals analysis

For heavy metal analysis, seagrass samples underwent chemical preparation for Atomic Absorption Spectrometry (AAS) (Abdallah et al. 2024). The elements detected in this study included Fe, Mn, Zn, Cu, Cr, Co, Cd, Ni, and Pb. Sample preparation involved dissolving approximately 0.70 g of dried seagrass in 10 mL of concentrated nitric acid (HNO_3) in a fume hood to ensure safe handling and minimize vapor exposure. The mixture was heated for 15 minutes, reducing the volume to about 3 mL. Subsequently, 20

mL of distilled water was added, and the sample was filtered to remove insoluble material. The filtered solution was then transferred to a 100 mL test tube for further analysis.

2.2.3 DNA damage analysis

DNA damage in seagrass samples was assessed following a previously described procedure. Briefly, five biological replicates from each sampling site were analyzed using the comet assay. Leaf tissues (1 cm^2) were placed in 90 mm Petri dishes, kept on ice, and finely chopped in 75 mM Sörensen buffer (pH 6.8) supplemented with Na_2EDTA and DMSO. An aliquot of 80 µL of the resulting nuclear suspension was gently mixed with 1% low–melting-point agarose pre-warmed to 40 °C, and 30 µL of the mixture was pipetted onto microscope slides pre-coated with 1% normal–melting-point agarose. Slides were incubated in lysis buffer (3 M NaCl, 150 mM Na_2EDTA, 1% Triton X-100, 10% DMSO, 15 mM Tris, pH 10) for 90 min at 4 °C, followed by incubation at 37 °C for 20 min. Subsequently, slides were washed three times with 0.5 M Tris-HCl (pH 7.5) and immersed in cold alkaline solution (1 mM Na_2EDTA, 250 mM NaOH, pH > 13) at 4 °C for 60 min.

Then, slides were placed in a horizontal electrophoresis system with the electrophoresis buffer (300 mM NaOH, 1 mM Na_2EDTA, pH 13) for 30 min. Electrophoresis was then carried out for 40 min at 0.8 V/cm with 300 mA and 4 °C. Neutralization was performed using 0.3 M Tris-HCl (pH 7.5) for three cycles of 10 min each, after which the slides were air-dried for 12 hours. Slides were then stained with SYBR Gold for 45 min and rinsed with distilled water for 5 min. DNA damage was evaluated using a fluorescence microscope at 200X magnification. Image J program (National Institute of Health, Bethesda, MD, USA) was used to quantify DNA comet length by selecting randomly at least 60 nuclei per slide. Five independent slides were evaluated per collection site.

2.3 Equations and Calculations

Equation (1) was applied to calculate the activity concentration (A) in Bq/kg of the detected radionuclides. Each radionuclide was identified using its characteristic energy line(s).

$$A = \frac{N}{\varepsilon \bullet m \bullet P_\gamma} \qquad (1)$$

Where N represents the counting rate estimated under the photopeak for each energy line. ε is the efficiency and m is the mass of the sample. P_γ represents the gamma emission probability for each energy line. In the present study the following spectral energy lines (% yield) were used to determine the detected radionuclides; 1460.80 (10.67%) keV for ^{40}K and the average value of the activity concentrations for ^{214}Bi and ^{214}Pb to estimate ^{226}Ra. ^{214}Bi was calculated using the average of its spectral lines with the highest yield (609.31 (46.1%), 1120.29 (15.1%), and 1764.49 (15.4%)) keV. And ^{214}Pb was obtained from the average of the three most abundant lines (242.00 (7.43%), 295.22 (19.3%), and 351.93 (37.6%) keV. ^{228}Ra was calculated from ^{228}Ac energy lines (338.32 (11.4%), 911.20 (26.2%), and 968.96 (15.9%)) keV. ^{238}U was calculated from its daughter; ^{234}Th at 63.29 (2.18%) keV spectral line and ^{232}Th was detected through its daughters ^{208}TI (583.0 keV) & ^{212}Pb (238.62 keV). Finally, the 46.54 (4.25%) keV energy line was used for ^{210}Pb (Bé & Chechev 2013).

The average values of the minimum detectable activities (MDA) of the detected radionuclides (^{40}K, ^{226}Ra, ^{228}Ra, ^{238}U, and ^{210}Pb) in the present study were 70.9, 4.2, 5.3, 15.9, and 27.8 Bq/kg, respectively.

Several radiological indices were calculated to assess the radiological hazards associated with naturally occurring radionuclides, including the radium equivalent activity (Ra_{eq}), the total absorbed dose rate (DR), and the external hazard index (H_{ex}).

The radium equivalent activity (Ra_{eq}) is widely used to evaluate the combined radiological effect of the activity concentrations (A) of the radionuclides ^{226}Ra, ^{232}Th, and ^{40}K where 10 Bq/kg of 226Ra, 7 Bq/kg of 232Th, and 130 Bq/kg of 40K producing the same dose rate of gamma ray. As shown in equation (2) (UNSCEAR 2000):

$$Ra_{eq}\,(Bq/kg) = A_{Ra} + 1.43A_{Th} + 0.077\,A_K \qquad (2)$$

The following equation (3) represents the calculation of the total absorbed dose rate (D_R) in nGy/h (UNSCEAR 2000, UNSCEAR 2008):

$$D_R\,(nGy/h) = 0.429A_{Ra} + 0.662A_{Th} + 0.0427A_K \qquad (3)$$

Where A_{Ra}, A_{Th}, and A_K represent the activity concentrations (in Bq/kg) of ^{226}Ra, ^{232}Th, and ^{40}K, respectively. The constants 0.429, 0.662, and 0.0427 are the conversion factors needed to determine the absorbed gamma dose rate per unit activity (nGy/h per Bq/kg).

The external hazard index (H_{ex}), which accounts for gamma radiation emitted from the samples, was used to evaluate the potential radiological risk to humans. It is calculated using equation (4) (UNSCEAR 2000):

$$H_{ex} = (A_U/370) + (A_{Th}/259) + (A_K/4810) \quad (4)$$

The primary purpose of this index is to ensure that the annual effective dose does not exceed the recommended safety limit which should be less than unity to be negligible.

The heavy metal concentrations in the seagrass samples were calculated using equation (5):

$$\text{A (ppm)} = \frac{C_T \times V_S \times \text{DF}}{W_S} \quad (5)$$

Where C_T is the concentration of the trace element in (AAS), V_S is the volume of the solution, DF represents the dilution factor with typical value of 1, and W_S is the sample weight.

2.4 Statistical analysis

Five replicates were used for each site for all experiments. Means were compared using one way ANOVA. Mean comparison was performed using Tukey's significant difference test. A significance level of 5% was used for all statistical analysis.

3. RESULTS AND DISCUSSION

3.1 Radiation Analysis

The measured activity concentrations of the detected radionuclides were presented in Fig. 2. We should note here that ^{232}Th was not detected in any of the samples. The ^{40}K, ^{226}Ra, and ^{238}U radionuclides were detected at all sites. Site SG2 showed the lowest activity concentrations for most radionuclides except for ^{238}U, which was close to the overall mean value.

The average ^{40}K activity concentration was 337.1±80.0 Bq/kg with values ranging between 85.4±6.5 Bq/kg at site SG2 in *Halophila stipulacea* sample and 694.1±19.9 Bq/kg at site SG3 in *Halophila stipulacea* sample. Potassium is normally accumulated in seagrasses through biological processes and is essential for growth (Peterson et al. 2007). The results obtained were consistent with other studies, in which the levels of ^{40}K were reported to range from 299.1 Bq/kg to 861.8 Bq/kg (Al-Absi et al. 2016), but were lower than those reported by Khandaker et al. (2019), who observed higher values ranging from 2200 to 3800 Bq/kg.

The mean activity concentration of ^{226}Ra radionuclide was 47.9±28.2 Bq/kg with a minimum value of 8.6±0.6 Bq/kg at site SG2 in *Halophila stipulacea* sample and a maximum value 216.3±3.3 Bq/kg at site SG5 in *Halophila stipulacea* sample. The ^{238}U activity concentration ranged from 25.5±1.7 Bq/kg at site SG5 in *Halophila stipulacea* sample to 65.5±7.1 Bq/kg at site SG1 in *Halodule uninervis*, with an average value of 45.8±5.1 Bq/kg. These results were comparable with previous findings from the same region, where ^{226}Ra activity concentrations were reported between <42.5 and <72.9 Bq/kg (Al-Absi et al. 2016). ^{228}Ra was detected at all sites except samples collected from the site SG2 in *Halophila stipulacea* sample; where it was below the minimum detectable limit. The highest activity concentration was 28.6±1.8 Bq/kg at site SG1 in *Halophila stipulacea* sample, with an average value 20.3±4.2 Bq/kg. ^{210}Pb was detected only at sites SG2 in *Halodule uninervis* sample and SG3 in *Halophila stipulacea* sample with values of 110.7±31.1 Bq/kg and 76.1±18.3 Bq/kg, respectively.

Two different types of seagrass (*Halophila stipulacea* &*Halodule uninervis*) were collected from sites SG1 and SG2 for comparison purposes. The results showed that the activity concentrations of all radionuclides were similar for both seagrass types except for the ^{40}K and ^{210}Pb activity concentrations. *Halophila stipulacea* showed higher activity concentrations at site SG1, while *Halodule uninervis* showed higher values at site SG2. This result is expected because the two species differ in their ability to take up radionuclides from sediments through their root systems and from seawater through their leaf tissue, as well as their physiological and morphological charachteristics that affect radionuclide accumulation in their tissues.

The sites were chosen carefully to compare the relatively undisturbed area, represented by SG2, and other sites with intensive human activities, especially near industrial areas SG1, SG3, SG4, and SG5. The results were generally consistent with this expectation, as the minimum activity concentrations were found in the green area, except for ^{40}K and ^{210}Pb concentrations in some samples.

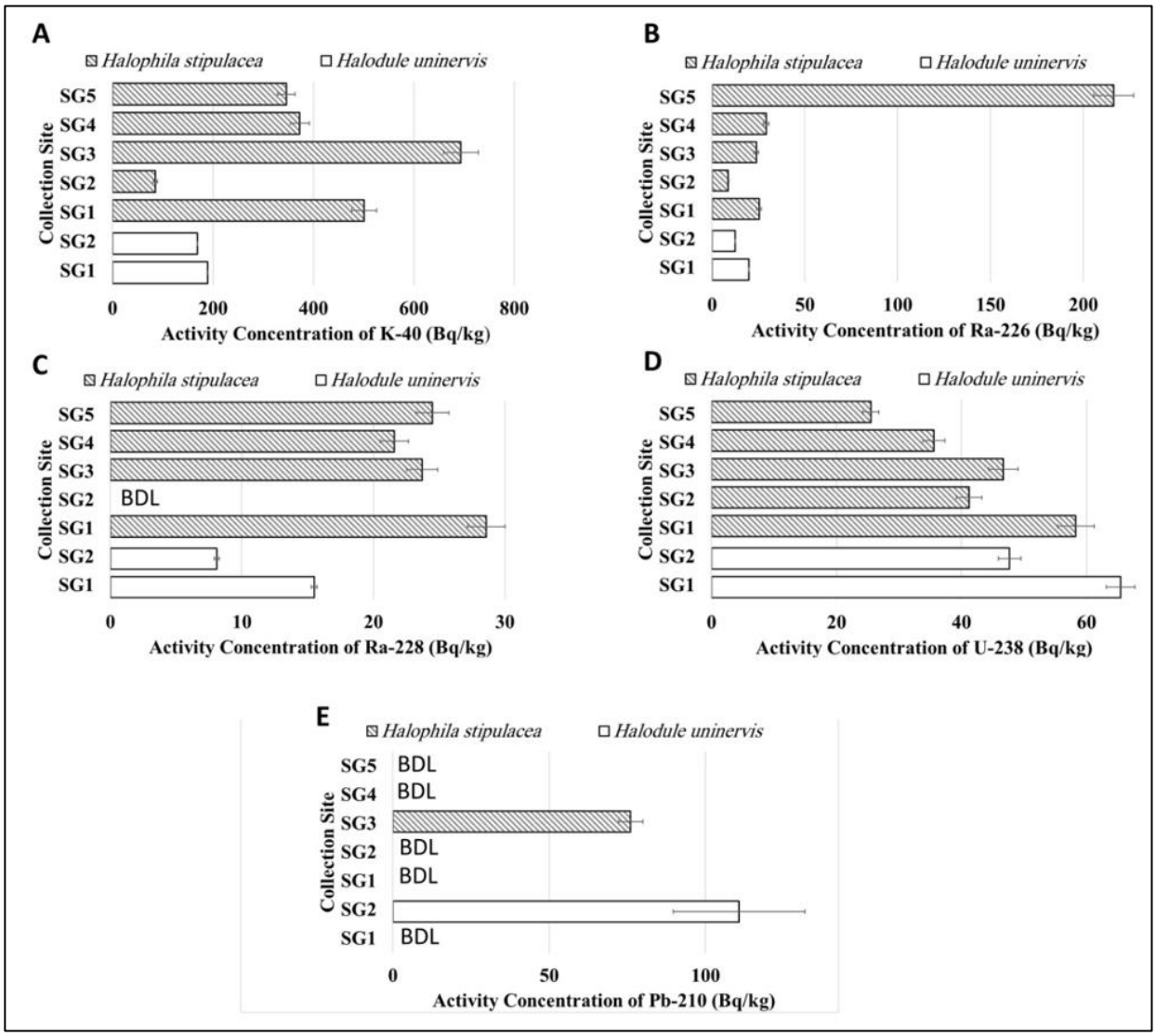

Fig. 2: Activity concentrations (Bq/kg) of ^{40}K, ^{226}Ra, ^{228}Ra, ^{238}U, ^{210}Pb in seagrass samples collected from the Gulf of Aqaba, Jordan. BDL: below detectable limit.

Radium equivalent (Ra_{eq}), the total absorbed dose rate (D_R), and the external hazard index (H_{ex}) were calculated and listed in Table 2 using equations 2, 3, & 4. The Ra_{eq} values in all samples were below the recommended limit of 370 Bq/kg (UNSCEAR 2000) as shown in Fig.3, suggesting that they are unlikely to pose significant radiological risks under normal exposure conditions. A similar trend in Fig. 3 was observed for the D_R with one sample (SG5) showing elevated value compared with others and exceeded the recommended limit of 55 nGy/h (UNSCEAR 2000). The results reflect the local differences between

different sites. The calculated values of H_{ex} were all below the recommended value, which should be unity (UNSCEAR 2000). Overall, the obtained results indicate that the radiological status of the samples investigated is within acceptable levels, although the site (SG5) exhibiting comparatively high value should continue to be monitored to assess the potential long-term exposure.

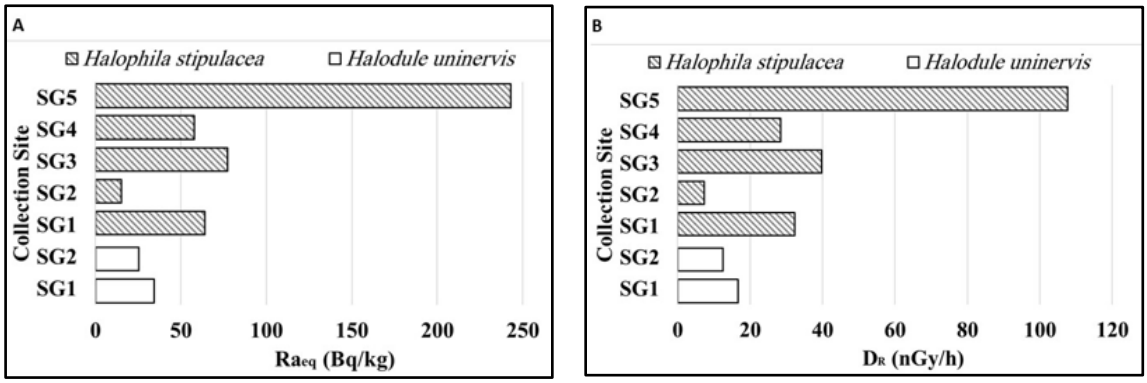

Fig 3: Radium Equivalent Ra_{eq} (Bq/kg) and Dose Rate D_R (nGy/h) in seagrass samples collected from the Gulf of Aqaba, Jordan.

3.2 Heavy Metal Analysis

The seagrass samples were analyzed for heavy metals, including Fe, Mn, Zn, Cu, Cr, Co, Cd, Ni, and Pb. However, the concentrations of Cu, Cr, Co, Cd, Ni, and Pb were below the detection limits of the AAS, where the values of detection limits (in ppm) for these elements were 0.382±0.001, 0.682±0.002, 0.309±0.001, 0.627±0.002, 1.70±0.003, and 1.60 ±0.005, respectively. Calibration was performed using certified multi-element standard solutions, and calibration curves exhibited correlation coefficients greater than 0.999. Quality-control procedures included reagent blanks, duplicate analyses, and spike-recovery tests. Recovery values ranged from 88% to 103%, while blank concentrations remained below the corresponding detection limits.

As a result, only the concentrations of Fe, Mn, and Zn are shown in Fig. 4. The occurrence of these heavy metals is consistent with other studies from the Red Sea and Arabian Gulf (Abdallah et al. 2005, Al-Adilah et al. 2021), also the measured concentrations were compared with background concentrations reported for seagrasses and marine macrophytes in the literature rather than with regulatory food-safety

limits, as no universally accepted permissible limits exist for seagrass tissues (Lewis & Devereux 2009, Sánchez-Quiles et al. 2017).

The concentrations of Zn ranged between 0.088±0.001 ppm and 0.929±0.002 ppm with average value of 0.436±0.002 ppm. Mn levels ranged between 0.470±0.001 ppm and 0.718±0.003 ppm with an average value of 0.597±0.002 ppm. The concentrations of both elements were within permissible limits at all sites.

Fe concentration results ranged from 5.775±0.005 ppm to 45.079±0.005 ppm (average 21.834 ppm). Levels were within permissible limits <30 ppm (Markert 1992), at the sites SG1 (*Halodule uninervis)*, SG2 for both (*Halodule uninervis & Halophila stipulacea)*, and SG5 but exceeded limits at the sites SG3 (31.968±0.008 ppm) and SG4 (31.238±0.012 ppm), indicating contamination. At the site SG1 (*Halophila stipulacea*) showed the highest concentration (45.079±0.005 ppm).

The concentrations of the heavy metals measured in the present work were substantially lower than those reported in another study from Cox's Bazar coast, Bangladesh (Bhuyan et al. 2024). Despite the low concentration observed in the collected samples, continued monitoring is necessary because even low-level contamination may contribute to cumulative ecological effects.

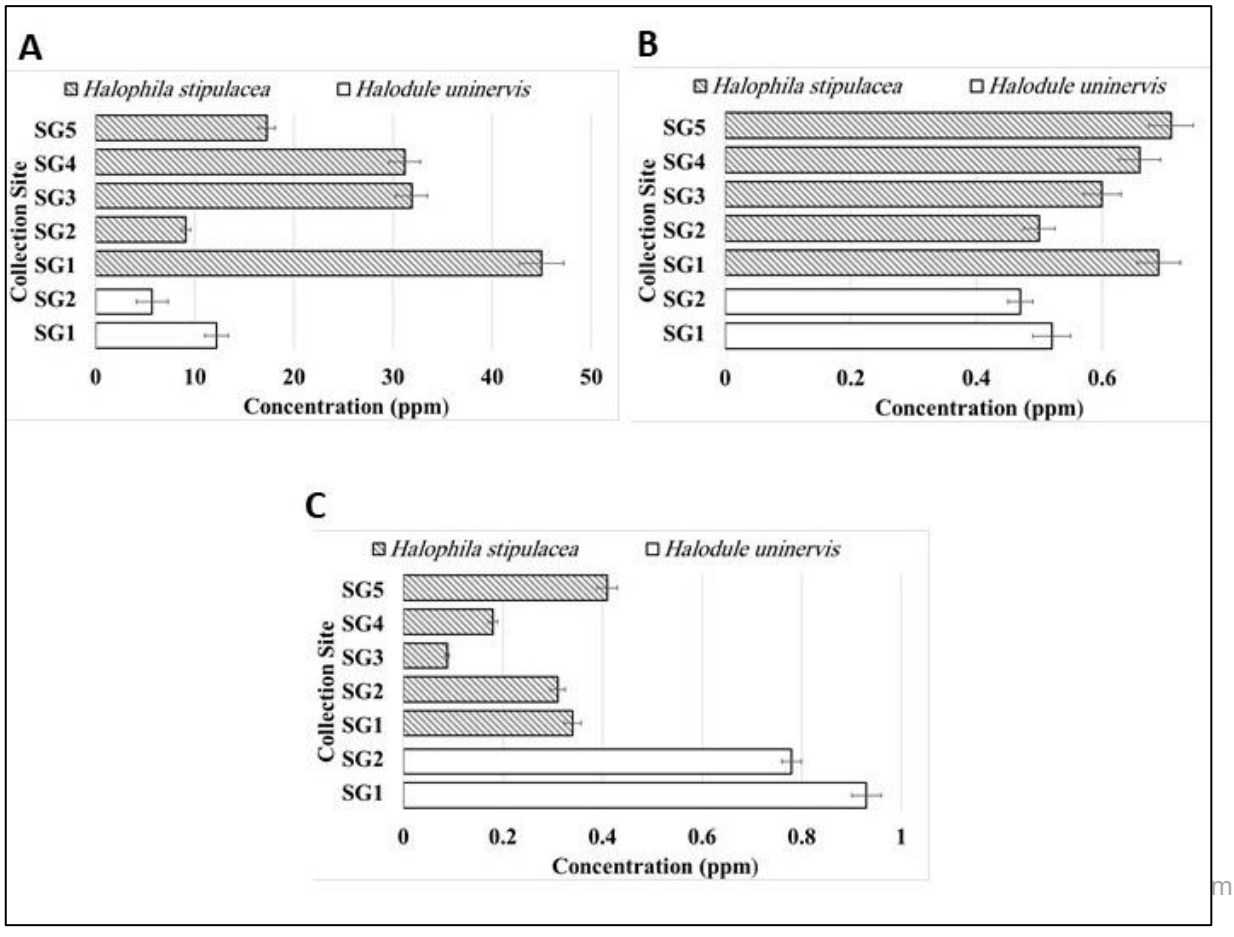

Fig. 4: Concentrations of heavy metals in seagrass samples collected from the Gulf of Aqaba, Jordan: (A) Fe, (B) Mn, and (C) Zn. Error bars represent standard error

3.3 DNA Damage Analysis

In this study, the comet assay was used to assess DNA damage in two seagrass species collected from five different sites in the Gulf of Aqaba, Jordan. Comet tail lengths measured in seagrass samples were in the range of 1.3±0.1 to 6.2±0.3 µm, with a mean value of 3.2±0.2 µm (Fig. 5). Similar tail lengths were observed in *Halodule uninervis* samples collected from SG1 and SG2. On the other hand, a significant difference in comet tail length was observed in *Halophila stipulacea* samples collected from different sites. Samples collected from SG4 and SG5 showed the longest tails, measuring 5.8±0.3 and 6.2±0.3 µm, respectively. Samples collected from SG3 showed an intermediate tail length of 4.1±0.2 µm.

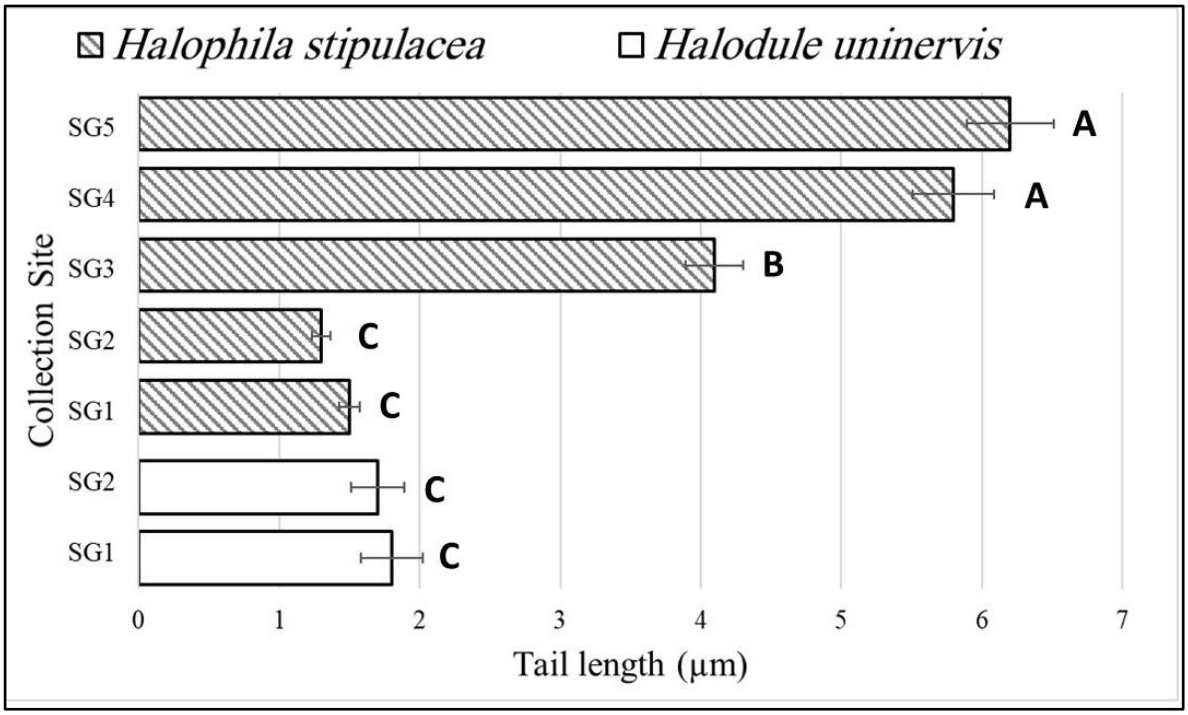

Fig 5: Comet tail length (µm) in seagrass samples collected from different sites in the Gulf of Aqaba-Jordan. Error bars represent standard error. Means followed by different letters are significantly different at P ≤0.05 according to Tukey's test.

The reported concentrations of radionuclides and heavy metals in the present study showed species-specific differences in element accumulation, which may be related to differences in structure, chemistry, and physiology (Jaiswar et al. 2025, Fievet et al. 2023). Since seagrasses do not have the same root-based

uptake pattern as terrestrial plants, they take up substances directly from seawater across their leaf surfaces, but the way this happens varies among species and types of elements. Some substances are absorbed passively by binding to components of the plant cell wall, while others enter cells through active transport processes that require energy. Studies have shown, for example, that brown algae accumulate radioactive cesium differently from red and green species due to differences in ion transport mechanisms (Davis et al. 2003).

The correlation analysis presented in Tables 2 and 3 showed no clear relationship between the measured DNA damage parameter and the concentrations of heavy metals in the studied samples, as indicated by the low correlation coefficients. In contrast, DNA damage showed weak positive correlations with both radium equivalent activity (Ra_{eq}) of 0.73 and absorbed dose rate (D_R) of 0.74. Although these correlations suggest a possible contribution of radiation exposure to the observed genetic damage, their relatively low magnitude indicates that the effect is limited and cannot solely explain the variability in DNA damage among the samples. The absence of a significant association with heavy metal concentrations further suggests that the measured metal levels were not the primary factor influencing genotoxic responses in the investigated organisms. These findings suggest that additional environmental factors, biological variability, or the combined influence of multiple stressors may contribute to the observed levels of DNA damage.

Table 2: Correlation coefficients (r) between DNA damage parameter and Radium equivalent & Dose Rate in the studied samples.

Site / Sample	DNA	Activity Concentration (Bq/kg)					H_{ex}	Ra_{eq} (Bq/kg)	D_R (nGy/h)
	Tail length (µm)	^{40}K	^{226}Ra	^{228}Ra	^{238}U	^{210}Pb			
SG1 *(Hu)*	1.8±0.1	190.1±22.3	19.8±1.6	15.5±1.8	65.5±7.1	BDL	0.09	34.4	16.6
SG2 (*Hu)*	1.7±0.1	169.0±11.6	12.4±1.0	8.1±1.3	47.7±7.2	110.7±31.1	0.07	25.4	12.5
SG1 (*Hs*)	1.5±0.1	501.6±23.7	25.4±1.3	28.6±1.8	58.3±5.7	BDL	0.17	64.0	32.3
SG2 (*Hs*)	1.3±0.1	85.4±6.5	8.6±0.6	BDL	41.2±4.6	BDL	0.04	15.2	7.3
SG3 (*Hs*)	4.1±0.2	694.1±19.9	23.8±0.8	23.7±1.0	46.7±4.5	76.1±18.3	0.21	77.2	39.8
SG4 (*Hs*)	5.8±0.3	373.6±24.2	29.1±1.6	21.6±1.8	35.6±7.0	BDL	0.16	57.9	28.4
SG5 (*Hs*)	6.2±0.3	346.3±19.3	216.3±3.3	24.5±1.6	25.5±1.7	BDL	0.66	243.0	107.6
Correlation (r)								0.73	0.74

**Hu: Halodule uninervis* **Hs: Halophila stipulacea*

Table 3: Correlation coefficients (r) between DNA damage parameter and concentration of heavy metals in the studied samples.

Site / Sample	DNA	Concentration (ppm)		
	Tail length (µm)	Fe	Mn	Zn
SG1 *(Hu)*	1.8±0.1	12.229±0.002	0.525±0.001	0.929±0.002
SG2 (*Hu)*	1.7±0.1	5.775±0.005	0.470±0.001	0.780±0.002
SG1 (*Hs*)	1.5±0.1	45.079±0.005	0.691±0.003	0.344±0.001
SG2 (*Hs*)	1.3±0.1	9.167±0.005	0.506±0.001	0.317±0.002
SG3 (*Hs*)	4.1±0.2	31.968±0.008	0.608±0.001	0.088±0.001
SG4 (*Hs*)	5.8±0.3	31.238±0.012	0.663±0.002	0.185±0.001
SG5 (*Hs*)	6.2±0.3	17.384±0.011	0.718±0.003	0.412±0.001
Correlation (r)		0.40	0.51	-0.57

**Hu: Halodule uninervis*
**Hs: Halophila stipulacea*

4. CONCLUSIONS

Seagrass samples were collected from five different sites in the Gulf of Aqaba, Jordan to assess the effect of radionuclide activity concentrations and heavy metal contamination on DNA damage. The radioactivity results showed the presence of five radionuclides (^{40}K, ^{226}Ra, ^{228}Ra, ^{238}U, and ^{210}Pb) across the sampled sites. The obtained values varied according to the seagrass species and the sampling location. The calculated values of radium equivalent

activity (Raeq), the total absorbed dose rate (DR), and the external hazard index (Hex) were all below the internationally recommended limits, , except at one site (SG5), suggesting negligible radiological risk and compliance with radiation safety standards. The analysis of heavy metal levels (Fe, Mn, Zn, Cu, Cr, Co, Cd, Ni, and Pb) in seagrass showed that most samples had concentrations within permissible limits, except for iron, which exceeded the permissible limit at selected sites. The DNA damage assessment showed measurable DNA damage in the collected samples. These findings indicate that DNA damage in the studied samples cannot be explained solely by heavy metals or radiological parameters. The weak correlations with radium equivalent activity and dose rate, along with the absence of a relationship with heavy metals, suggest that other environmental and biological factors may contribute to the observed DNA damage. Further studies are needed to better understand the factors influencing genotoxic effects in the investigated ecosystem.

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