THE UNIVERSITY OF YAOUNDE I
UNIVERSITE DE YAOUNDE I

FACULTY OF SCIENCE
FACULTE DES SCIENCES

CENTRE FOR RESEARCH AND DOCTORAL TRAINING IN GRADUATE STUDIES IN LIFE SCIENCE, HEALTH AND ENVIRONMENT
CENTRE DE RECHERCHE DE FORMATION DOCTORALE SCIENCE DE LA VIE, SANTE ET ENVIRONNEMENT

UNIT FOR RESEARCH AND DOCTORAL TRAINING IN GRADUATE STUDIES IN HEALTH AND ENVIRONMENT
UNITE DE RECHERCHE SANTE ET ENVIRONNEMENT

DEPARTMENT OF PLANT BIOLOGY
DEPARTEMENT DE BIOLOGIE ET PHYSIOLOGIE VEGETALES

Assessment of heavy metal bioaccumulation capacities of some lowland plants in Yaounde (Cameroon)

Dissertation Defended for the Award of a Doctorate/Ph. D in Plant Biology

Option: Plant Biotechnologies
Speciality: Environmental Sanitation

Presented by:

AYO Anne
Master ès Sciences
Mat: 06Q061

Director :
KENGNE NOUMSI Ives Magloire †
Associate Professor

Supervisor :
DJOCGOUE Pierre François
Professor

Year 2021

UNIVERSITE DE YAOUNDE I
THE UNIVERSITY OF YAOUNDE I

FACULTE DES SCIENCES
FACULTY OF SCIENCE

DEPARTEMENT DE BIOLOGIE ET PHYSIOLOGIE VEGETALES
DEPARTMENT OF PLANT BIOLOGY

Yaoundé, le 0 8 FEV 2022

ATTESTATION DE CORRECTION

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Nous soussignés, membres du jury de soutenance de la thèse de **Doctorat/Ph. D** en **Biologie et Physiologie Végétales**, soutenue le vendredi **21 janvier 2022** par Madame **AYO Anne**, Master ès Science, Matricule **06Q061**, intitulée « ***Assessment of heavy metal bioaccumulation capacities of some lowland plants in Yaounde (Cameroon)*** **»**, certifions qu'elle a effectué les corrections conformément aux remarques et recommandations du jury.

En foi de quoi, nous lui délivrons cette attestation de correction pour servir et valoir ce que de droit. /-

Jury de Soutenance

Le Président

Youmbi Emmanuel
Professeur

Rapporteur

Pierre François Djocgoue
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Membre N°1

AMBANG Zachée
Professeur

Membre N°2

Pr Léopold Tagouet Serge H.
Université de Yaoundé I

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LISTE DES ENSEIGNANTS PERMANENTS	**LIST OF PERMANENT TEACHING STAFF**

ANNÉE ACADEMIQUE 2020/2021

(Par Département et par Grade)

DATE D'ACTUALISATION: 22 Septembre 2021

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4	KANSCI Germain	Professeur	En poste
5	MBACHAM FON Wilfried	Professeur	En poste
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10	NJAMEN Dieudonné	Professeur	En poste
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12	NOLA Moïse	Professeur	En poste
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15	ZEBAZE TOGOUET Serge Hubert	Professeur	En poste
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30	KANDEDA KAVAYE Antoine	Chargé de Cours	En poste
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35	NGOUATEU KENFACK Omer Bébé	Chargé de Cours	En poste
36	NGUEMBOK	Chargé de Cours	En poste
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39	TADU Zephyrin	Chargé de Cours	En poste
40	TAMSA ARFAO Antoine	Chargé de Cours	En poste
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45	FEUGANG YOUMSSI François	Assistant	En poste
46	FOKAM Alvine C. Epse KEGNE	Assistant	En poste
47	GONWOUO NONO Legrand	Assistant	En poste
48	KOGA MANG DOBARA	Assistant	En poste
49	LEME BANOCK Lucie	Assistante	En poste
50	NWANE Philippe Bienvenu	Assistant	En poste
51	YOUNOUSSA LAME	Assistant	En poste
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20	MAHBOU SOMO TOUKAM. Gabriel	Chargé de Cours	En poste
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23	KOUOTOU DAOUDA	Chargé de Cours	En poste
24	MAKON Thomas Beauregard	Chargé de Cours	En poste
25	MBEY Jean Aime	Chargé de Cours	En poste
26	NCHIMI NONO KATIA	Chargée de Cours	En poste
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16	MKOUNGA Pierre	Maître de Conférences	En poste
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30	NGOMO Orléans	Chargée de Cours	En poste
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32	SIELINOU TEDJON Valérie	Chargé de Cours	En poste
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36	OUETE NANTCHOUANG J. Laure	Assistant	En poste
37	TCHAMGOUE Joseph	Assistant	En poste
38	TSAFFACK Maurice	Assistant	En poste
39	TSAMO TONTSA Armelle	Assistant	En poste
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6	DJAM Xaviera YOUHEP KIMBI	Chargé de Cours	En Poste

7	DOMGA KOMGUEM Rodrigue	Chargé de Cours	En Poste
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19	EKODECK Stéphane Gaël Raymond	Assistant	En poste
20	HAMZA Adamou	Assistant	En poste
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22	MAKEMBE. S. Oswald	Assistant	En poste
23	MESSI NGUELE Thomas	Assistant	En poste
24	MEYEMDOU Nadège Sylvianne	Assistante	En poste
25	NKONDOCK. MI. BAHANACK.N.	Assistant	En poste
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15	DOUANLA YONTA Herman	Chargé de Cours	En poste
16	FOMEKONG Christophe	Chargé de Cours	En poste
17	KIKI Maxime Armand	Chargé de Cours	En poste
18	MBAKOP Guy Merlin	Chargé de Cours	En poste
19	MENGUE MENGUE David Joe	Chargé de Cours	En poste
20	NGUEFACK Bernard	Chargé de Cours	En poste
21	NIMPA PEFOUNKEU Romain	Chargée de Cours	En poste
22	POLA DOUNDOU Emmanuel	Chargé de Cours	En poste
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24	TCHANGANG Roger Duclos	Chargé de Cours	En poste
25	TETSADJIO TCHILEPECK M. E.	Chargée de Cours	En poste

26	TIAYA TSAGUE N. Anne-Marie	Chargée de Cours	En poste
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33	OGADOA AMASSAYOGA	Assistante	En poste
34	TCHEUTIA Daniel Duviol	Assistant	En poste
35	TENKEU JEUFACK Yannick Léa	Assistante	En poste
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8	SADO KAMDEM Sylvain Leroy	Maître de Conférences	En poste
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10	BOUGNOM Blaise Pascal	Chargé de Cours	En poste
11	ESSONO OBOUGOU Germain G.	Chargé de Cours	En poste
12	NJIKI BIKOÏ Jacky	Chargée de Cours	En poste
13	TCHIKOUA Roger	Chargé de Cours	En poste
14	ESSONO Damien Marie	Assistant	En poste
15	LAMYE Glory MOH	Assistante	En poste
16	MEYIN A EBONG Solange	Assistant	En poste
17	MONI NDEDI Esther Del Florence	Assistant	En poste
18	NKOUDOU ZE Nardis	Assistant	En poste
19	SAKE NGANE Carole Stéphanie	Assistante	En poste
20	TAMATCHO KWEYANG Blandine Pulchérie	Assistant	En poste
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5	KOFANE Timoléon Crépin	Professeur	En poste
6	NANA ENGO Serge Guy	Professeur	En poste
7	NANA NBENDJO Blaise	Professeur	En poste
8	NDJAKA Jean Marie Bienvenu	Professeur	*Chef de Département*
9	NJANDJOCK NOUCK Philippe	Professeur	En poste
10	NOUAYOU Robert	Professeur	En poste
11	PEMHA Elkana	Professeur	En poste
12	TABOD Charles TABOD	Professeur	*Doyen FS Univ./Bda*

13	TCHAWOUA Clément	Professeur	En poste
14	WOAFO Paul	Professeur	En poste
15	ZEKENG Serge Sylvain	Professeur	En poste
16	BIYA MOTTO Frédéric	Maître de Conférences	*DG/HYDRO Mekin*
17	BODO Bertrand	Maître de Conférences	En poste
18	ENYEGUE A NYAM épse BELINGA	Maître de Conférences	En poste
19	EYEBE FOUDA Jean sire	Maître de Conférences	En poste
20	FEWO Serge Ibraïd	Maître de Conférences	En poste
21	HONA Jacques	Maître de Conférences	En poste
22	MBANE BIOUELE César	Maître de Conférences	En poste
23	MBINACK Clément	Maître de Conférences	En poste
24	NDOP Joseph	Maître de Conférences	En poste
25	SAIDOU	Maître de Conférences	*Chef de centre/ IRGM/MINRESI*
26	SIEWE SIEWE Martin	Maître de Conférences	En poste
27	SIMO Elie	Maître de Conférences	En poste
28	VONDOU Derbetini Appolinaire	Maître de Conférences	En poste
29	WAKATA née BEYA Annie	Maître de Conférences	*Directeur/ENS/UYI*
30	ABDOURAHIMI	Chargé de Cours	En poste
31	CHAMANI Roméo	Chargé de Cours	En poste
32	EDONGUE HERVAIS	Chargé de Cours	En poste
33	FOUEDJIO David	Chargé de Cours	*Chef Cell. MINADER*
34	MBONO SAMBA Yves Christian U.	Chargé de Cours	En poste
35	MELI'I Joelle Larissa	Chargée de Cours	En poste
36	MVOGO ALAIN	Chargé de Cours	En poste
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38	WOULACHE Rosalie Laure	Chargée de Cours	En poste
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44	WANDJI NYAMSI William	Assistant	En poste
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11	GANNO Sylvestre	Maître de Conférences	En poste
12	GHOGOMU Richard TANWI	Maître de Conférences	*CD/UMa*
13	MOUNDI Amidou	Maître de Conférences	*CT/MINIMDT*
14	NGUEUTCHOUA Gabriel	Maître de Conférences	*CEA/MINRESI*
15	NJILAH Isaac KONFOR	Maître de Conférences	En poste
16	NYECK Bruno	Maître de Conférences	En poste
17	ONANA Vincent Laurent	Maître de Conférences	*Chef service Maintenance & du Matériel/UYII*
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19	TCHOUANKOUE Jean-Pierre	Maître de Conférences	En poste
20	TEMDJIM Robert	Maître de Conférences	En poste
21	YENE ATANGANA Joseph Q.	Maître de Conférences	*Chef Div. /MINTP*
22	ZO'O ZAME Philémon	Maître de Conférences	*DG/ART*
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24	BEKOA Etienne	Chargé de Cours	En poste
25	ELISE SABABA	Chargé de Cours	En poste
26	ESSONO Jean	Chargé de Cours	En poste
27	EYONG JOHN TAKEM	Chargé de Cours	En poste
28	FUH Calistus Gentry	Chargé de Cours	*Sec. D'Etat/MINMIDT*
29	LAMILEN BILLA Daniel	Chargé de Cours	En poste
30	MBESSE CECILE OLIVE	Chargée de Cours	En poste
31	MBIDA YEM	Chargé de Cours	En poste
32	METANG Victor	Chargé de Cours	En poste
33	MINYEM Dieudonné-Lucien	Chargé de Cours	*CD/UMa*
34	NGO BELNOUN Rose Noël	Chargée de Cours	En poste
35	NGO BIDJECK Louise Marie	Chargée de Cours	En poste
36	NOMO NEGUE Emmanuel	Chargé de Cours	En poste
37	NTSAMA ATANGANA Jacqueline	Chargé de Cours	En poste
38	TCHAPTCHET TCHATO De P.	Chargé de Cours	En poste
39	TEHNA Nathanaël	Chargé de Cours	En poste
40	TEMGA Jean Pierre	Chargé de Cours	En poste
41	FEUMBA Roger	Assistant	En poste
42	MBANGA NYOBE Jules	Assistant	En poste

Répartition chiffrée des enseignants permanents par Département

Département	Nombre d'enseignants				
	Pr	MC	CC	ASS	Total
BCH	8 (1)	14 (9)	13 (5)	05(2)	**40 (18)**
BPA	15 (1)	8 (6)	18 (5)	10 (3)	**51 (15)**
BPV	7 (1)	9 (1)	9 (6)	7 (1)	**32 (10)**
C.I.	10 (1)	9 (2)	10 (2)	3 (0)	**32 (5)**
C.O.	6 (0)	21 (5)	5 (2)	8 (2)	**40 (9)**
IN	2 (0)	1 (0)	14 (1)	8 (1)	**25 (2)**
MAT	2 (0)	8 (0)	15 (1)	9 (2)	**34 (7)**
MIB	3 (0)	5 (3)	6 (1)	6 (2)	**20 (6)**
PHY	15 (0)	14 (2)	9 (3)	8 (3)	**46 (8)**
ST	7 (1)	15 (1)	18 (5)	2 (0)	**42 (7)**
Total	**75 (5)**	**104 (30)**	**116 (31)**	**66 (16)**	**361 (86)**

Soit un total de 361 (86) dont :

- Professeurs	**75 (5)**
- Maîtres de Conférences	**104 (30)**
- Chargés de Cours	**116 (31)**
- Assistants	**66 (16)**
- ( ) = Nombre de femmes	**86**

DEDICATION

To

My late Director Professor Ives Magloire KENGNE NOUMSI

My parents Mr. and Mme NGUEWO

My dear sister Miss MBEDA Myrene

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- the members of the jury, who despite their occupation, have accepted to examine this work. I am sure their suggestions will help upgrade its scientific value;
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SUMMARY

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ABREVIATIONS

ATSDR: Agency for Toxic Substances and Disease Registry

BAF: Bioaccumulation factor

CEC: Cation Exchange Capacity

DS: Dry season

MTEs: Metal Trace Elements

HM: Heavy metals

ICP-OES: Inductively Coupled Plasma - Optical Emission Spectroscopy

Igeo: geo-accumulation index

IITA: International Institute of Transforming Agriculture

IPI: Integrated Pollution Index

MR: Mobility ratio

OC: Organic Carbon

OM: Organic matter

PAHs: Polynuclear Aromatic Hydrocarbons

PEL: Probable effect concentration

pH: potential Hydrogen

PI: Pollution index

ROS: Reactive Oxygen Species

RS: Rainy season

SDG: Sustainable Development Goals

SQG: Sediment Quality Guidelines

TF: Translocation factor

TMRPL: Total Metal Removal Pollution Load

USDA: United States Department of Agriculture

WHO: World Health Organization

WWTS: Wastewater treatment system

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ABSTRACT

Lowlands are very important ecosystems, offering commendable goods and services to humans and the environment. However, they could be very fragile. Lowlands in Yaounde are subject to soil pollution due to various wastes, particularly heavy metals from anthropogenic activities. The aim of this study is to assess the bioaccumulation capacities of heavy metals of some plant species growing in lowlands of Yaounde in Cameroon. In this study, the identification of macrophytes that thrive on heavy metals polluted soils remains an essential step in the phytoremediation process. Floristic inventory was conducted using the quadrat method, in order to identify macrophyte species with phytoremediation capacities. Twelve lowlands were selected, of which eleven were polluted and one was the control (unpolluted area). Soil, water and plant samples were collected during the dry and rainy seasons in three potentially polluted lowlands of Yaounde namely: Mokolo-elobi (site 4), Mvan (site 9) and Etang Atemengue Obili (site 11), and physico-chemical parameters of soil and water samples were determined. The concentrations of Pb, Cd, Cu, Ni, Zn, Cr, As and Co were determined in three soil and water samples, and twenty-four plant organ samples to assess the level of metal pollution. Pollution and geoaccumulation indices as well as ecological risks were used to assess the level of soil contamination. Similarly, in water, the toxicity level, pollutant load and ecological risks of metals were assessed using the pollution, metal assessment and toxicity load indices. Principal component analyses (PCA) and cluster analyses (CA) were used to determine the metals sources in soil and water. The remediation performances of plants were determined using the mobility ratio, translocation (roots - aerial parts) and bioaccumulation factors. The metal accumulation and bioconcentration indices in plants were analysed.

During the rainy season, 189 species belonging to 138 genera distributed in 63 families were identified in polluted sites, while 139 species belonging to 103 genera dispatched in 39 families were identified in the dry season. Macrophyte diversity was higher in each polluted site compared to the control. The Poaceae, Asteraceae, Fabaceae, Malvaceae, and Solanaceae families showed a higher taxonomic richness on polluted sites in the rainy season (23, 20, 14, 12 and 12 taxa respectively), compared to the Poaceae, Asteraceae, Cyperaceae, Convolvulaceae and Fabaceae families (19, 17, 9, 8 and 8 species respectively) in the dry season. The results revealed a high diversity of species present in polluted lowlands, with Shannon's diversity index (H'=2.63) and Pielou's equitability index (J'=0.459-0.847), as compared to the control (H'=2.34 and J'=0.747) in the rainy season. The respective values were H'=2.61 and J'=0.692-0.819 as compared to the control (H'=2.45 and J'=0.866) in the dry season. Based on the characteristic criteria of accumulating species and their metal accumulation capacities, plants were classified into major relative frequence and relative abundance of the species (Fri and A> 10%) and intermediate (Fri > 10% and 2%≤ A< 10) categories. Thus, 15 assorted species had presented interesting characteristics that could be tested in preliminary trials to investigate their phytoremediation/cleaning-up capacities, among which *Echinochloa pyramidalis*, *Pennisetum purpureum* and *Commelina benghalensis* were chosen. Concerning soils, the mean concentrations (n=3) of Cr (202.01±83.81 µg/g) in three sites, Ni (80.29±24.88 µg/g) at site 11 and Co (8.17±0.6; 20.23±1.7 µg/g) at sites 11 and 4 of the study were high as compared to the threshold limits for soils used for irrigation agriculture. The geoaccumulation index (Igeo) values indicated that soils were heavily contaminated by Cr and moderately by Cu from anthropogenic sources. The Nemerow integrated pollution index (IPI) revealed the pollution of all 3 soils by heavy metals and classified it as follows: site 11 (8.06) > site 9 (5.79) > site 4 (3.41). The potential ecological risks (E^i_r) of toxic metals followed the order of Cr>Cu>Co>Pb>Ni>As>Zn>Cd and indicated a slight level of ecological risk, with Cr and Cu being the highest contributors to the increase of the ecological risk level in the lowlands. During the dry season, the mean concentrations (n=3) of Cd (0.336±0.235 mg/L), As (0.335±0.236 mg/L) and Co (0.34±0.235 mg/L) in water were higher than the standard used for irrigation agriculture. Total heavy metal toxicity load and heavy metal evaluation index values were found to be lower than the acceptable value. According to the ecological risk index classification, 100% of the total samples were found to pose low ecological risk during both seasons. The mean concentration of heavy metals in plants species (n=8) of Pb (9.67±6.05 µg/g), Cd (0.41±0.38 µg/g), Cr (22.36±17.09 µg/g), Ni (7.63±5.88 µg/g), Zn (252.62±65.71 µg/g), Cu (25.92±1.82 µg/g), As (0.00±0.00 µg/g) and Co (4.69±4.23 µg/g) were all above the norm except for As. *C. benghalensis* presented the highest accumulation of Zn. The metal accumulation gradient in plant organs followed a decreasing order, from roots>leaves>stems, with the exception Cd (stems>roots>leaves) and Zn (roots>stems>leaves), thus showing an antagonism in the uptake of these two metals by plants. *C. benghalenis* has the potential to be used as phytoextractor of Zn, Cu and Cd (translocation factor (TF) and bioaccumulation factor (BAF)>1) while *E. pyramidalis* can be used for Cd and Ni, and *P. purpureum* for Cd, but they are all phytostabilizers of Co and As. Thus, the heavy metals translocation from soil to plants characterized them as indicators of metal pollution. The present study revealed a high accumulation of metals in the Poaceae family due to their highly developed, fibrous, extensive and diversified root systems compared to the Commelinaceae, explainaing their rapid development in contaminated lowland areas.

Keywords: Heavy metals, bioaccumulation capacities, lowland, plants, Yaounde.

RESUMÉ

Les bas-fonds sont des écosystèmes très importants, qui fournissent des biens et services louables à l'homme et à l'environnement. Pourtant, ils peuvent être très fragiles. Les bas-fonds de Yaoundé sont soumis à la pollution due aux déchets de toutes sortes, notamment les métaux lourds provenant des activités anthropiques. L'objectif de cette étude était d'évaluer les capacités bioaccumulatrices des métaux lourds chez quelques plantes se developpant dans les bas-fonds de Yaoundé au Cameroun. Au cours de cette étude, l'identification des macrophytes capables de se développer dans les bas-fonds pollués par des métaux lourds est restée une étape essentielle dans le processus de phytoremédiation. L'inventaire floristique a été réalisée grâce à la méthode de quadrats afin d'identifier les espèces de plantes ayant des capacités phytoremédiatrices. Pour cela, 12 bas-fonds ont été sélectionnés dont 11 contaminés et 01 témoin (non pollué). Des échantillons de sols, d'eaux et de plantes ont été prélevés pendant les saisons sèche et pluvieuse dans trois bas-fonds sélectionnés potentiellement pollués: Mokolo-elobi (site 4), Mvan (site 9) et Etang Atemengue Obili (site 11), et les paramètres physico-chimiques des échantillons de sols et d'eaux ont été déterminés. Les concentrations de Pb, Cd, Cu, Ni, Zn, Cr, As et Co ont été déterminées dans 30 échantillons pour évaluer le niveau de pollution des métaux dans le sol, l'eau et les plantes. Les indices de pollution (PI), de géoaccumulation (Igeo), ainsi que le risque écologique (E^{i}_{r}) ont été utilisés pour évaluer le niveau de contamination des sols. De même, dans l'eau, le niveau de toxicité, la charge polluante et le risque écologique des métaux ont été évalués en utilisant les indices de pollution, d'évaluation des métaux et de charge de toxicité. Les analyses en composantes principales (PCA) et clusters (HA) ont permis de déterminer les sources des métaux dans le sol et l'eau. Les capacités phytoremédiatrices des plantes ont été déterminées en utilisant le ratio de mobilité, les facteurs de translocation (racines - parties aériennes) et de bioaccumulation. Les indices d'accumulation et de bioconcentration des métaux dans les plantes ont été analysés.

Pendant la saison des pluies, 189 espèces appartenant à 138 genres et réparties dans 63 familles ont été identifiées sur les sites contaminés, tandis que 139 espèces appartenant à 103 genres et réparties dans 39 familles ont été identifiées en saison sèche. La diversité des macrophytes était plus élevée dans chaque site pollué comparé au témoin. Les familles des Poaceae, Asteraceae, Fabaceae, Malvaceae, et Solanaceae ont montré une plus grande richesse taxonomique sur les sites pollués en saison des pluies (23, 20, 14, 12 et 12 taxons respectivement), comparé aux familles des Poaceae, Asteraceae, Cyperaceae, Convolvulaceae et Fabaceae (respectivement 19, 17, 9, 8 et 8 espèces) pendant la saison sèche. Les résultats ont révelé une grande diversité des espèces présentes dans les bas-fonds pollués avec les indices de diversité de Shannon (H'=2,63) et d'équitabilité de Pielou (J'= 0,459-0,847), comparés au temoin (H'=2.34 et J'=0.747) en saison des pluies. Les valeurs respectives étaient H'=2,61 et J'=0,692-0,819 comparés au temoin (H'=2.45 et J'=0.866) pendant la saison sèche. Sur la base des critères caractéristiques des espèces accumulatrices et leurs capacités d'accumulation des métaux, les résultats ont permis de classer les plantes en catégories majeures, fréquence et abondance relatives (Fri et A)> 10% et intermédiaires (Fri > 10% et 2%≤ A< 10). Ainsi, 15 espèces assorties ont présenté des caractéristiques intéressantes pouvant faire l'objet d'essais préliminaires pour étudier leurs capacités de phytoremédiation, parmi lesquelles *Echinochloa pyramidalis, Pennisetum purpureum* et *Commelina benghalensis* ont été retenues. Concernant les sols, les concentrations moyennes (n=3) de Cr (202,01±83,81 µg/g) dans les 3 sites, de Ni (80,29±24,88 µg/g) dans le site 11 et de Co (8,17±0,6; 20,23±1,7 µg/g) dans les sites 11 et 4 des bas-fonds étudiés étaient élevées comparées aux seuils admissibles pour les sols utilisés pour l'agriculture. Les valeurs de l'indice de géoaccumulation (Igeo) ont indiqué que les sols étaient fortement contaminés par le Cr et modérément par le Cu provenant des sources anthropiques. L'indice de pollution intégré (IPI) de Nemerow a revélé la pollution des sols dans les 3 sites par les métaux lourds et les a classés comme suit: site 11 (8,06)>site 9 (5,79)>site 4 (3,41). Le risque écologique potentiel (E^{i}_{r}) des métaux toxiques suivait l'ordre de Cr>Cu>Co>Pb>Ni>As>Zn>Cd et indiquait un léger niveau de risque écologique, le Cr et le Cu contribuaient le plus à l'augmentation du niveau de risque écologique dans les bas-fonds. Pendant la saison sèche, les concentrations moyennes (n=3) de Cd (0,336±0,235 mg/L), As (0,335±0,236 mg/L) et Co (0,34±0,235 mg/L) dans l'eau étaient supérieures aux normes de rejet dans l'environnement. La charge totale de toxicité et les valeurs de l'indice d'évaluation des métaux lourds se sont avérées inférieures aux valeurs acceptables. Selon la classification de l'indice de risque écologique, 100 % des échantillons totaux se sont avérés présenter un faible risque écologique pendant les deux saisons. Les concentrations moyennes (n=8) des métaux lourds dans les plantes Pb (9,67±6,05 µg/g), Cd (0,41±0,38 µg/g), Cr (22,36±17,09 µg/g), Ni (7,63±5.88 µg/g), Zn (252,62±65,71 µg/g), Cu (25,92±1,82 µg/g), As (0,00±0,00 µg/g) et Co (4,69±4,23 µg/g) étaient tous au dessus des normes, sauf pour l'arsenic. *C. benghalensis* a présenté la plus forte accumulation de Zn. Le gradient d'accumulation des métaux dans les organes de la plante a suivi un ordre décroissant, de racines>feuilles>tiges, à l'exception du Cd (tiges>racines>feuilles) et du Zn (racines>tiges>feuilles), montrant un antagonisme dans l'absorption de ces 2 métaux par les plantes. *C. benghalenis* a montré son potentiel pour être utilisé comme phytoextracteur de Zn, Cu et Cd (facteur de translocation (TF) et de bioaccumulation (BAF)>1) tandis que *E. pyramidalis* pour Cd et Ni, et *P. purpureum* pour Cd, mais elles sont toutes des phytostabilisateurs de Co et As. Ainsi, la translocation des métaux du sol vers les plantes, les caractérises comme indicateurs de pollution métallique. La présente étude a révélé une forte accumulation des métaux dans la famille des Poaceae dues à leurs systèmes racinaires très développés, fibreux, étendus et diversifiés comparé aux Commelinaceae, expliquant leur développement rapide dans les zones contaminées des bas-fonds.

Mots clés: Métaux lourds, capacités bioaccumulatrices, bas-fonds, plantes, Yaoundé.

CHAPTER I. GENERALITIES

I.1. Introduction

I.1.1. Context

Considered as fertile wetlands and objects of good agricultural covetousness, lowlands present important planning and sanitation challenges. Lowlands are primary landscape elements, but their presence contributes to the fragmentation of urban space, creating a challenge for town planning. According to the Ramsar convention, wetlands are generally areas of marsh fen, peat land or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water, the depth of which at low tide does not exceed six meters (Anonymous 1, 2013).

Cameroon is richly endowed with large numbers of lowlands characterized by a lagoon system (Chebo, 2009). Lowlands are very important subsystems of the general ecosystem as they play vital roles in the sustenance of terrestrial, surface and groundwater resources. Lowlands are important through their values and functions. In Yaounde, lowlands contribute to food production especially vegetables. They are also involved in ecological balance and biodiversity. Lowland functions include flood control, groundwater recharge, coastal protection, sediment traps, atmospheric equilibrium, waste treatment, and they also provide nurseries for aquatic life (Anonymous 1, 2013; Moomaw et *al*., 2018). Increasingly colonized by housing, agricultural and market gardening activities associated with breeding and fish farming, lowlands are useful for the creation of green corridors within the city. However, these activities tend to provoke some serious problems in the ecosystem. In addition, lowlands of Yaounde are increasingly becoming receptacles for more or less controlled domestic waste and other types of more toxic waste such as used oils, sewage sludge and hydrocarbons.

The accumulation of heavy metals in soils impacts agricultural production because of their harmful effects on crop growth, food quality and environmental health (Emurotu & Onianwa, 2017; Kumar et *al*., 2020; Proshad et *al*., 2020). Heavy metals are serious environmental pollutants, especially in areas with high anthropogenic pressure (Briffa et *al*., 2020). Their presence in soil exhibits toxic effects towards soil biota by affecting key microbial processes and decrease the number of microorganisms and their activities in the soil. Even low concentrations of heavy metals may inhibit the physiological metabolism of plants. However, unlike other pollutants such as hydrocarbons, household and municipal wastes that can accumulate openly in the environment, trace metals can accumulate and reach toxic levels without being noticed. Their transfer in the food chain represents a significant public health risk because of the pathologies they can cause. Despite efforts to identify and estimate the extent of wetland pollution around the world, the lack of comprehensive data is an obstacle to the mobilization of economic resources that can help to minimize soil pollution. However, maintaining the health of lowland areas, preventing and reducing soil and water pollution is possible through the promotion of sustainable wetland management practices, reduction, recycling and sustainable storage of waste, and environmentally sound industrial processes such as phytoremediation.

In Yaounde the capital of Cameroon, the trend is the same as in most African cities. The growing population estimated at 2,395,583 inhabitants in 2015 with an annual population growth rate of 2.8% (Anonymous 9, 2012), exerts considerable pressure on the environment. The increased rate at which effluents are discharged into the environment is as a result of rapid urbanization, characterized by high population growth with inadequate urban planning capabilities. Consequently, the management of wastes in marshy lowlands is limited due to lack of funding, a deficit in organizational actions, inadequacy of collection equipment and a lack of urban plan (Sotamenou, 2012; Kumar et *al*., 2017; Almuktar et *al*., 2018). However, it should be noted that big cities generate large volumes of effluent loaded with toxic substances, especially heavy metals originating from industrial activities (Emurotu & Onianwa, 2017; Adesuyi et *al*., 2018; Kumar et *al*., 2020). These effluents, being dumped hazardously in nature, are leached and accumulate in lowlands, thereby increasing the eutrophication level of these ecosystems.

The remediation of contaminated environments is essential and research continues to develop new technologies that rely on the phytoremediation capacities of plants. Increasingly expensive physical techniques, such as chemical inactivation or landfill sequestration, are being replaced by biological methods based on scientific research like microbial degradation or phytoremediation (Singh et *al*., 2018; Rodríguez-Eugenio et *al*., 2018; Dhaliwal et *al*., 2020). Preventing lowland soil and water pollution could reduce land degradation, increase food security, contribute significantly to adaptation and mitigation of climate change, and help avoid conflicts and migration. Therefore, taking immediate action against soil pollution could help achieve almost all of the Sustainable Development Goals (SDGs), with a significant impact on Goals 1, 2, 3, 6, 9, 11, 12, 13, 14, 14, 15 and 17.

Despite the enormous scientific progress made to date, the protection and monitoring of soil conditions at the national and global levels still face complex challenges that hinder effective policy and decision-making on the ground. In recent decades, interest was focused on phytoremediation technology which is the exploitation of the capacity of plants to solve the problems of high pollution loads and anoxia or hypoxia (Yan et *al*., 2020; Zhang et *al*., 2020). Studies demonstrated the role played by several species of lowland plants in water purification (Adesuyi et *al*., 2018; Ite & Ibok, 2019).

Soils and water pollution are major public health concern in popular towns worldwide (Jamal et *al*., 2018). Air or water pollution differs from soil polluted by metals because in soil, heavy metals persist much longer than in other compartments of the biosphere (Briffa et *al*., 2020). Due to an increase in population density in urban areas, soil quality is strongly influenced by anthropogenic activities (industrial and socio-economic activities) and then differs significantly from natural soils (Biyogue, 2016; Wanjala et *al*., 2019). A report on the State of the World's Soil Resources (Anonymous 2, 2019) has identified the contamination and pollution of soils by heavy metals as one of the main threats to soils and ecosystem services. Many publications have also reported a great deal on human health problems linked to contamination by metallic trace elements (Kumar et *al*., 2017; Singh et *al*., 2018).

Despite the relatively low level of industrial activity in less developed regions like Africa, there is a high potential of toxic pollution by heavy metals. In recent decades, the rise of industrial and construction activities has exponentially increased the amount of waste (solid, liquid and gaseous vapor), thereby increasing the penetration of heavy metals into the environment. Across the African continent, this situation seems to have increased the concentration of target heavy metals such as cadmium (Cd), lead (Pb), zinc (Zn), nickel (Ni), copper (Cu), chromium (Cr), cobalt (Co) and arsenic (As) in the environment. Indeed, average background concentrations of 6×10^{-2} ppm for Cd, 14.53 ppm for Cu, 2.88 ppm for Zn and 17.69 ppm for Pb were found in water from the Olezoa wetland (Fonkou et *al*., 2005). Defo et *al*. (2015) presented the geo-accumulation indices for Pb (0.13 - 0.19), Cr (0.13 - 0.16) and Ni (0.09 - 0.11) in the Ntem basin, indicating the contamination of soils by these metals in the urban watershed of Yaounde. Heavy metal contamination of the lowland area causes a severe environmental problem. Therefore, soils accumulate different types of pollutants from non-point and/or point of sources of pollution (Chen et *al*., 2017; Yang et *al*., 2020).

Phytoremediation is a process that involves the use of plants to extract and or sequester pollutants in order to detoxify a medium. It is considered as an effective, low-cost, long-lasting, esthetic solution and a preferred clean-up option for contaminated areas (Singh et *al*., 2018; Yan et *al*., 2020). However, macrophytes growing in lowland areas are exposed to many sources of contamination. Anthropogenic sources include industrial effluents such as effluents from mining extractions, petrochemical activities, seepage of oil wells and oil tankers, oil spills on crop farms, areas occupied by flow stations, oil wells, gas flaring sites, pipeline laying sites, burrow pits, and other oil exploration and exploitation activities. (Mandeng et *al*., 2019; Obasi et *al*., 2019). Other potential sources of heavy metals in the environment include sewage from household and hospital effluents, urban runoff, metal scrubbing, burning of fossil fuels (automobile exhaust) (Agoro et *al*., 2020) and incineration of solid wastes and agricultural effluents (metals contained in pesticides) (Ali et *al*., 2018; Zhou et *al*., 2020).

I.1.2. Problem statement

Rapid urbanization and industrialization have led to an increase in pollution from landfill leachates, industrial effluents, vehicle emissions (gas from exhaust pipes), fertilizer erosion as agricultural run-off, herbicides and pesticides, sewage and wastewater. All these contribute to the accumulation of pollutants in nearby aquatic systems (Adesuyi et *al*., 2015; Adesuyi et *al*., 2016). Among the worst environmental contaminants are the heavy metals contaminants (Kamari et *al*.,

2017) indicating the contamination of soils by metals in the urban watershed of Yaounde (Defo et *al*., 2015). They are serious pollutants due to their toxicity, persistence in natural conditions and ability to be incorporated into food chains (Adesuyi et *al*., 2015; Masindi & Muedi, 2018). Some of these metals includes lead (Pb), cadmium (Cd) copper (Cu), zinc (Zn), nickel (Ni), chromium (Cr), cobalt (Co) and arsenic (As). In fact, the subject of phytoremediation of heavy metals has been studied all over the world. However, there is limited data on chemical pollution for soil, water and plants in tropical Africa and Cameroon in particular. A number of issues related to heavy metals pollution need to be understood and investigated. In Cameroon, many authors have focused their investigations on urban wastewater (sewage from households and runoff), constructed wetlands, industrial effluents and agricultural wastes (Kengne et *al*., 2008; Fonkou et *al*., 2010; Soh et *al*., 2014). According to Datta et *al*. (2020), wetland ecosystems must possess water, plants and soils. The plants of such ecosystems should adapt to life under saturated conditions. The macrophytic flora of lowlands in Yaounde city is diversified. In the Mfoundi division, lowlands are influenced by pollution. Tchinda et *al*. (2018) found that species like *Pennisetum purpureum*, *Commelina benghalensis* and *Echinochloa pyramidalis* were dominant. Fonkou et *al*. (2005) identified species like *Cyperus papyrus*, *Enhydra fluctuans*, *Ipomoea aquatica* and *Echinochloa pypramidalis* in the Olezoa basin flow in Yaounde, which were the most abundant and presented higher levels of heavy metal accumulation. Other relatively abundant species such as *Commelina nudiflora*, *Pteris atroviriens*, *Leersia hexandra* were also found in these wetlands, but the bioaccumulative capacities of these plants are neither determined nor evaluated. In effect, the lowland ecotechnology is not well known in the tropics due to the lack of information and local expertise in developing countries. These plants enable the direct assessment of the response of lowland vegetation to changes in aquatic discharge. In the past few decades, the use of the leaves of higher plants as biomonitors for heavy metal pollution has been on the increase especially in sensitive and urban areas (Shahid et *al*., 2017; Salih & Aziz, 2019). A study conducted by De Laet et *al*. (2019) demonstrated that *Eichhornia crassipes* was a powerful bioindicator for water pollution by emerging pollutants.

A good biomonitor will indicate the presence of the pollutants and also provide additional information about the quantity and intensity of the exposure (Alexandrino et *al*., 2020). Metal uptake by plants can be element specific, plant specie specific and plant tissue specific (Shahid et *al*., 2017; Kasowska et *al*., 2018). If the distribution of macrophytes in lowlands has been studied around the world by several previous researchers (Messou et *al*., 2013; Tchinda et *al*., 2018; Adesuyi et *al*., 2019), minimal efforts have been devoted to the bioaccumulation capacities and the occurrence of heavy metals in the lowland areas. Knowledge about the bioaccumulation of heavy metals in plant species of lowlands in Yaounde is limited. Hence, the aim of this study is to investigate the heavy metals accumulation capacities of some major plant species collected from lowlands in Yaounde in order to evaluate their potential for phytoremediation purposes.

I.1.3. Research questions

From this research, the following research questions were outlined:

- Which are the heavy metal pollution-tolerant plant species colonizing lowlands of Yaounde?
- What are the heavy metal contamination levels of soil, water and plants of lowland areas?
- What are the capabilities of plant species identified as bioaccumulators for the uptake heavy metals in polluted lowlands?

I.1.4. Hypothesis

- Plant species growing in lowlands are pollution-tolerant to heavy metal contamination.
- Soils, water and plants of lowland areas are polluted by heavy metals.
- Plant species mostly selected as bioaccumulators are efficient for heavy metals uptake in polluted lowlands.

I.1.5. Objectives of the research

The general objective of the present research is to evaluate the heavy metal bioaccumulation capacities of some plant species growing in the lowlands of Yaounde.

Specifically, the objectives are:

- to identify the pollution-tolerant macrophytic flora colonizing lowlands in Yaounde;
- to assess the level of heavy metals contamination in lowland soils, water and plants;
- to evaluate the heavy metal accumulation performances of three selected plant species mostly identified as bioaccumulators.

I.2. Literature review

I.2.1. Generalities on the heavy metals in soil and plant

I.2.1.1. Generalities on the heavy metals in soil

I.2.1.1.1. Definition of heavy metals

Conventionally, heavy metals (HMs) are defined as elements with metallic properties (ductility, conductivity, stability as cations etc.) with a density greater than 5 g/cm^3 (i.e., specific gravity greater than 5) and an atomic number = 20 (Tchounwou et *al*., 2014). There is not yet a consensus definition of the term "heavy metal" (HMs). Nevertheless, according to Duffus (2002); (Proshad et *al*., 2020), contamination or pollution and potential toxicity or ecotoxicity have often been associated with the term heavy as a group name for metals and semi-metals (metalloids). Recently, Ali & Khan (2018) proposed that heavy metals be defined as naturally occurring metals having atomic number greater than 20 and an elemental density greater than 5 g.cm^{-3}. In relation to biological and environmental studies, the term metals describes metallic elements with a potential toxic effect on living organisms even at very low concentrations. The term is used as follows: metal, metalloid, semi metal, light metal, heavy metal, essential metal, beneficial metal, toxic metal, abundant metal, available metal, trace metal and micronutrient (Duffus, 2002). The most common heavy metals present in contaminated soils are Cd, Pb, Hg, Cu, Cr, As and Zn (Masindi & Muedi, 2018; Zwolak & Sarzy, 2019).

I.2.1.1.2. Soil pollution and the origin of the contamination of lowlands by heavy metals

The release of metals into the environment is a consequence of a wide range of industrial activities and combustion of fossil fuels. These different sources contribute to the load of metal pollutants in aquatic and terrestrial food chains (Noubissie et *al*., 2016; Xu et *al*., 2017; Xu et *al*., 2018). As a result, human tissues may contain levels of metals above the standard (Guan et *al*., 2018). According to observations by Ferronato & Torretta (2019) and Korish & Attia (2020), the increase of anthropogenic influences accelerate the release of harmful wastes into the environment and these hazardous materials are released into the soil. Domestic wastewater can contain heavy metals in large quantities, making this source unsuitable (Milik & Pasela, 2018; Milik & Pasela, 2018). Many dangerous pollutants are part of these waters, which can have damaging effects on the ecosystem (Mahmoud & Ghoneim, 2016). In addition to other elements, another important source of Cd is the use of mineral phosphate fertilizers, which generally contain high concentrations of Zn and Cd.

Phosphate fertilizer application can directly increase Pb and Cd concentrations in the soil solution. The main sources of Pb pollution in agriculture and factories are lead mining, combustion of lead hydrocarbons, sewage sludge and farmyard manure spreading, industrial processes etc. (Nazarpour et *al*., 2019). The major sources of Pb in wetland soils can be attributed to industrial wastewater discharges, pesticides and fertilizer impurities, emissions from mining and smelting operations and atmospheric deposition from fossil fuel combustion.

Chromium (Cr) in soils could be due to the disposal of waste materials consisting of lead-chromium batteries, discarded plastics, colored polyethylene bags and empty paint containers (Amos-Tautua et *al*., 2014). In wetlands, the mobility of chromium depends on the sorption characteristics of the soil. These include clay content, iron oxide content and the amount of organic matter present. Chromium is transported by surface runoff to surface waters in its soluble or precipitated form (De Oliveira, 2019; Tripathi & Chaurasia, 2020). The main sources of soil contamination by nickel are metal plating industries, fossil fuel combustion, nickel mining and electroplating (Sevinç et *al*., 2014; Anum et *al*., 2019).

I.2.1.1.3. Transfer and availability of heavy metals in soil

Heavy metals are released to the soil and ground water due to anthropogenic activities. Industrial waste and sewage sludge disposals on land often contain significant amounts of heavy metals such as Cd, Ni, Zn, Cu, As, Cr, Co and Pb, which create a potential risk for the environment. Several studies have been focused on the retention behavior of heavy metals in soils (Qu et *al*., 2019; Maina et *al*., 2019; Maina et *al*., 2019). Physical and chemical processes govern heavy metal' behavior in soil.

The fate of MTEs will depend on various factors, such as the physico-chemical and biological parameters of the soil. These factors will control the processes of adsorption on the surfaces of solid particles (clays, hydroxides, organic matter), complexation with organic ligands, surface precipitation, ionic exchange or precipitation in the form of ion as salts or co-precipitation (Rahman et *al*., 2018). The fluxes of MTEs out of the soil are varied and quite difficult to assess. MTEs can reach surface or groundwater via lateral or vertical transfer (colloidal or soluble) in soils. They can be taken up by plants or soil organisms as well as by humans. Meso- and macrofauna also contribute to the transfer of trace metals (Allamin et *al*., 2020). It should be noted that there exist preferential circulation of water and colloids in soils in relation to soil porosity, root galleries or earthworms. Water or wind erosion also play an important role in the transfer of MTEs to the soil surface (Cui et *al*., 2019).

I.2.1.1.4. Natural and anthropogenic sources of heavy metals in the environment

I.2.1.1.4.1. Natural sources

Heavy metals in the environment can come from both natural (geogenic or lithogenic) and anthropogenic sources. Natural or geological sources of heavy metals in the environment result from weathering of metal-bearing rocks and volcanic eruptions. They are generally recovered from ores because of mineral processing (Zwolak & Sarzy, 2019). In rocks, heavy metals are found in different chemical forms of ores (such as sulfides of Pb, Co, Fe, As, Pb-Zn, Ag and Ni, and oxides of Se, Al, Mn and Sb) from which these metals are extracted as minerals (Shakoor et *al*., 2019). Consequently, these metals recovered/mined from soils as oxide and sulfide ores. Most of these heavy metals are obtained in the exhaust fumes in pyro metallurgical processes or as by-products of several hydro-metallurgical processes after mining. For example, Cd is mainly obtained as a by-product of Zn refining process, due to the presence of Cd in the sphalerite of the Zn ore. Each year, a significant amount of heavy metals is thus redistributed from the contaminated aquifer of earth's crust to different compartments of the environment, i.e. water, air and soil. As a result, soils derived from parent materials with a high concentration of metals in the bedrocks naturally have a high concentration of metals (Pourrut et *al*., 2011; Rodríguez-Eugenio et *al*., 2018; Pourret, 2019).

I.2.1.1.4.2. Anthropogenic sources

Anthropogenic sources of heavy metals in soil include: refining and mining of ores, pesticides, batteries, paper industries, tanneries, fertilizer industries, solid wastes disposal including sewage sludge, wastewater irrigation and vehicle exhaust (Shahid et *al*., 2015). Typically, heavy metals are released both as compounds (inorganic and organic) and as elements. Anthropogenic activities such as fertilizer leaching, inadequate disposal of industrial effluents, accidental oil spills, domestic sewage, minerals extraction and rainwater contaminated by heavy metals in the atmosphere are thought to contribute significantly to the pollution of aquatic ecosystem (Ferati et *al*., 2015; Adesuyi et *al*., 2018; Ali et *al*., 2020). In some cases, metals emitted by these processes continue to accumulate in soil and other environmental compartments, even long after the end of these activities. Anthropogenic sources of Cr include electroplating industries, leather tanneries, textile industries, and steel industries (Sharma et *al*., 2020). Wastewater can be divided into several categories, such as sanitary sewage, chemical wastewater, industrial mining wastewater and urban mining mixed sewage, etc. Heavy metals are brought to the soil by irrigation wastewater and are fixed in the soil in different ways. It causes the continuous accumulation of heavy metals (Hg, Cd, Pb, Cr, etc.) in soil year by year.

Heavy metals added to agricultural soils by inorganic fertilizers can leach into groundwater and contaminate it (Shahid et *al*., 2015). Phosphate fertilizers are particularly rich in toxic heavy metals. Fertilizers, pesticides and mulch are important agricultural inputs for crops production (Rai et *al*., 2019; Merga et *al*., 2020). Nevertheless, long-term over-application has led to soil contamination by heavy metals. The vast majority of pesticides are inorganic compounds or pure minerals, and some pesticides contain Hg, As, Cu, Zn and other heavy metals (Singh et *al*., 2017; Hassan, 2020). Fertilizers also generally contain high levels of Cr (Krüger et *al*., 2017). Anthropogenic increase in Cd concentrations are also due to the excessive application of chemical fertilizers (Wang et *al*., 2015). Phosphate fertilizers contain Cd as a contaminant at concentrations ranging from trace levels to 300 ppm dry weight and can therefore be a main source of this metal to agricultural systems (Gupta et *al*., 2014; Azzi et *al*., 2019). The heavy metal content of fertilizers is generally as follows: phosphoric fertilizer > compound fertilizer > potash fertilizer > nitrogen fertilizer (Gupta et *al*., 2014). In general, Pb is released into the environment from different sources including acid batteries, old plumbing systems, and lead shots used for hunting of game birds. The combustion of leaded gasoline is also a source of Pb in the environment.

Anthropogenic sources of heavy metals such as Pb come from waste incineration, vehicle exhaust, lead smelting and paint use are the main site of soil accumulation (Chrzan, 2015). The humic horizon of the soil is the place where the elements are strongly bounded and accumulated. Due to their low mobility, in acidic and sandy soils, Pb can become easily accessible to plants and, as a result, can be incorporated into food chains and pose a direct threat to organisms (Dinu et *al*., 2020; Liu et *al*., 2020). Organic matter generally strongly fixes the solubility of the element and this increases with soil acidity. Due to its emission in industrial dust, wastewater and sewage sludge fertilizers, soils become polluted with nickel. Zinc is a heavy metal commonly found in nature, accumulated in humus and permanently fixed in soil organic matter (Chrzan, 2015). The main sources of soil contamination by zinc are the zinc industries, the use of organic fertilizers and irrigation fields contaminated by municipal sewage and transport.

I.2.1.1.4.3. Typology of essential and non-essential heavy metals for living organisms

With respect to the role of heavy metals in living systems, they are classified into two groups:

- essential heavy metals are those, which are needed by living organisms and be required in low concentrations for their growth, development and physiological functions like Mn, Fe, Ni, Cu and Zn (Ali et *al*., 2019);
- non-essential or toxic heavy metals such as Cd, Pb, Hg and As are those, which are not needed by living organisms for any physiological function (Atobatele & Olutona, 2015; Hejna et *al*., 2018).

Higher levels of heavy metals disrupt the normal physiology and biochemistry of living systems. The most dangerous heavy metals are Pb, Hg, As, Cd, Sn, Cr, Zn and Cu (Engwa et *al*., 2019).

Among these, Cd and Pb are the most hazardous metals for human health, even at low concentrations (Arif et *al*., 2016; Masindi & Muedi, 2018). They are responsible for certain diseases and cause a number of human health risks (Sall et *al*., 2020). Moreover, they have been classified as carcinogenic for humans and wildlife (Kinuthia et *al*., 2020). For plants, heavy metals such as Co, Cu, Fe, Mn, Ni and Zn are essential micronutrients and Fe is required in the highest concentrations (Elmorsi et *al*., 2019). However, in excess they become toxic. Interactions between non-essential elements and essential micronutrients have been widely observed and reported. For instance, the absorption and translocation mechanisms of Fe in plants seem to be similar to those of Cr. Thus, non-essential elements are efficiently absorbed by the root and leaf systems (Ali et *al*., 2019).

I.2.1.2. Generalities on the heavy metals in plants

I.2.1.2.1. Transfer mechanism and metal elimination by plants

I.2.1.2.1.1. Plant physiology

Flowering plants or vascular plants are the most advanced plants (higher plants) with four different types of organs (roots, stems, leaves and flower) that ensure nutrition and reproduction (Garousi, 2017). The shape, size, colour and spatial organisation of these organs determine the plant's specific morphology. The different organs of the plant establish a network between water, soil and air and present a physiological functioning of the plant as well as an internal organisation allowing it to respond to its nutritional needs and environmental constraints (Dupuy, 2014). Leaves are the site of photosynthesis, which takes place in the chloroplasts. Solar energy is used to convert CO_2 and water into sugar following a series of reactions (photosynthesis, respiration and photorespiration (Fig. 1).

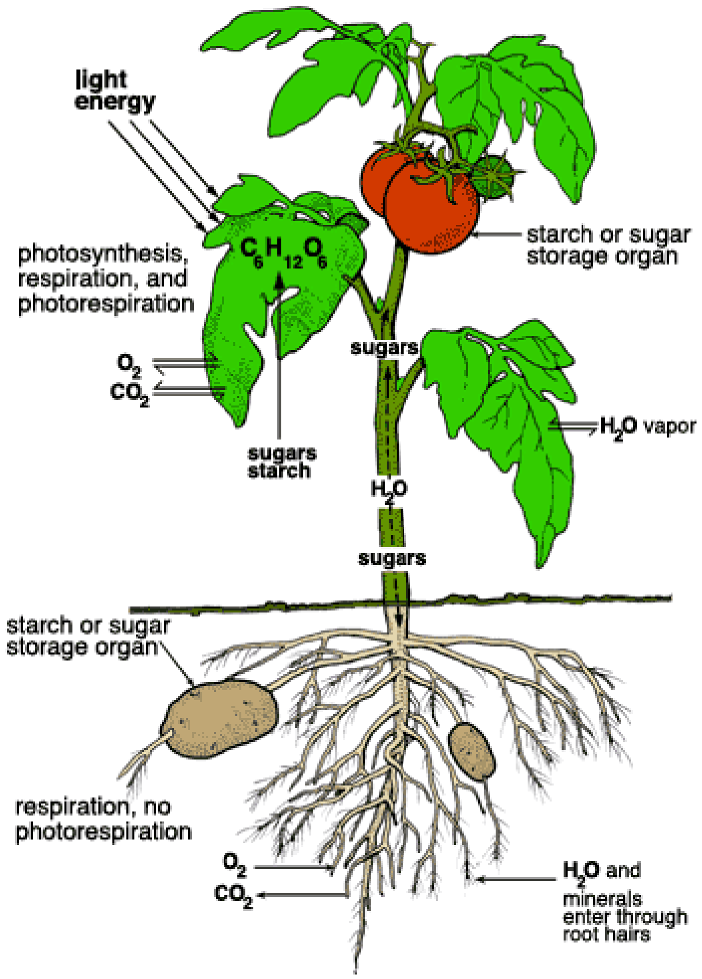

Fig. 1. Physiology and functioning of a higher plant (Garousi, 2017).

I.2.1.2.1.2. Ways of heavy metal entrance in plant organs

The entry of heavy metal into the plant depends both on the physico-chemical properties of these elements and on the physiological processes involved in the development of the plant

(Fernández et *al*., 2017). Before being taken up by plant, the pollutant is present in the soil in different forms: dissolved in the soil solution, volatilized or bound to the soil organic or mineral matrix. Only the phyto-available fraction can be taken up by plant during its growth. The main mechanisms of metal transfer within the plant or from the external environment to the plant are:

- uptake from the soil by the roots;
- uptake from the air by the leaves;
- diffusion through stomata and leaf cuticles from the gas phase of the air;
- translocation between plant compartments by xylem (root to leaf) and phloem (leaf to root) flow.

The mechanisms of uptake and accumulation of MTEs in plants can occur at different levels from uptake by the roots to accumulation in the leaves. Indeed, MTEs can enter the plant either by the roots or by the aerial pathways.

I.2.1.2.1.2.1. Root pathways

Root is an underground axis that grows downwards (positive gravitropism), and away from light (negative phototropism). Root is a non-chlorophyllous organ whose role is to fix the plant to the soil and to absorb water and minerals through its absorbent hairs. Overall, the roots of a plant are usually organised in three types of systems (Fig. 2):

- taproots: typical of dicotyledons, they are characterised by a large dominance of the seminal root over the lower roots (ie carrot (*Duacus carotta*));
- fasciculated roots: in plants where the seminal root emits secondary roots of approximately the same size as it, and especially when this root disappears in the profile of a bundle of lateral roots. This is observed in many monocotyledons such as grasses like *Cynodon dactylon*;
- adventitious roots: these roots originate on an aerial or underground stem or on any other part where the embryo dies quickly (ie maize (*Zea mays*)).

A root consists of three main parts: the meristematic zone, the elongation zone and the differentiation zone (Fig. 3). The absorbing hairs are the zone of maximum absorption through which water and solutes are transported via the xylem into the central stele. The main force that enables water extraction and conduction into the xylem is leaf transpiration and the tension it induces on the water column in the plant (Dupuy, 2014).

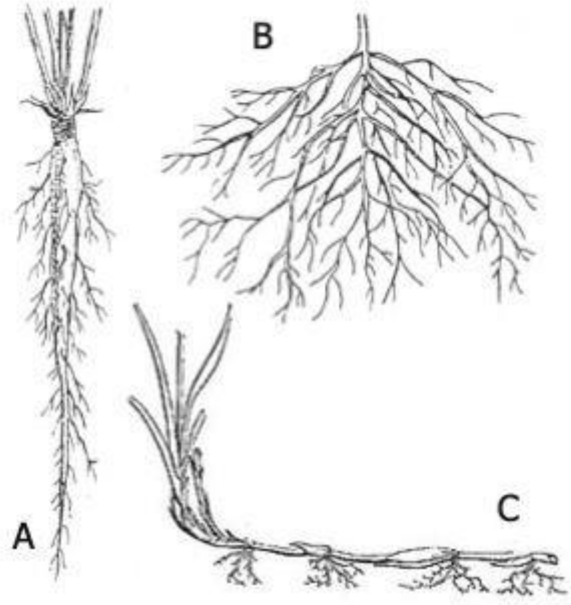

Fig. 2. Different root systems (A: taproots, B: fasciculated roots, C: adventitious roots).

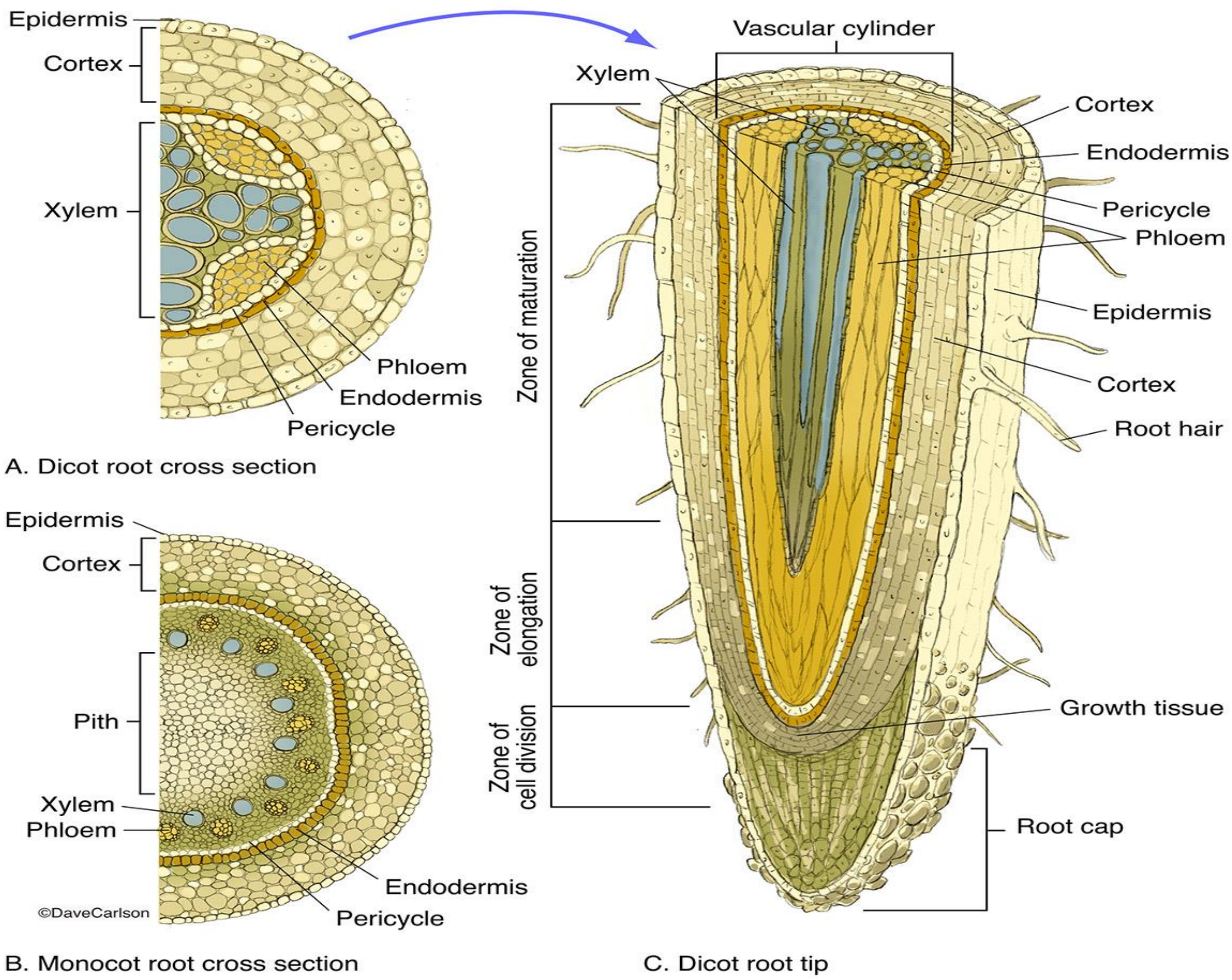

Fig. 3. Typical roots structure (Dupay, 2014)

The transfer of water and solutes from the soil solution takes place at the level of the non-lignified cells of the rhizodermis, transits through the cortical parenchyma to the endodermis, which due to its position and cellular characteristics, plays a fundamental role in the transit of water and mineral salts to the root absorption zone. Heavy metals are taken up in the same way as nutrients, which have similar properties (Hejna et *al*., 2018; Merga et *al*., 2020). Roots are able to produce peroxidase enzymes that oxidise metals. Roots release carbon because of tissue or cell loss and exudation of different types of compounds (mucilages, organic acids and minerals) (Bali et *al*., 2020). Most of the excreted molecules serve as a source of carbon and energy for the surrounding microorganisms and may therefore stimulate microbial activity in the rhizosphere.

It is mainly via the roots that hydromineral nutrition of plants takes place. Once the organic compound is dissolved in soil water, the absorption by the roots goes through a transfer of the molecules in the soil, then through a soil-plant transfer via the roots (Fig. 4). Thus, only the available fraction will be taken up by the plant during its growth. The molecules must first pass through the endodermis by the symplastic route in order to reach the xylem vessels. Only hydrophilic compounds will be able to pass through these symplasts while lipophilic compounds such as hydrocarbons are more likely to accumulate in the endodermis (Berardino, 2017). The function of the endoderm is to regulate the flow of substances between the ecorse and the conducting tissues. Once in the Caspari framework (a sub-permeable band of lignin), water and mineral salts that previously moved by the apoplastic route (between the pectocellulose walls), are forced to use the symplastic route (between the endodermal cells). Thus, the endodermal cell walls control the mineral elements moving in the radial direction, from the peel to the conducting tissues. However, the endodermis also regulates the outward flow of water and mineral elements (Dupuy, 2014).

Roots are also subject to the process of sorption of heavy metals. Sorption refers to the adsorption or absorption of a molecule onto or into another substance. It results from the action of gas or liquid molecules that have been exposed to a solid material and adhere to its surface (adsorption) or become incorporated into its entire volume (absorption). In both cases, the attached molecule is no longer present in the solvent.

The efficiency of heavy metal uptake at the root level can be assessed by the Root Concentration Factor (RCF), which determines its distribution between the roots and the surrounding medium (soil and/or water).

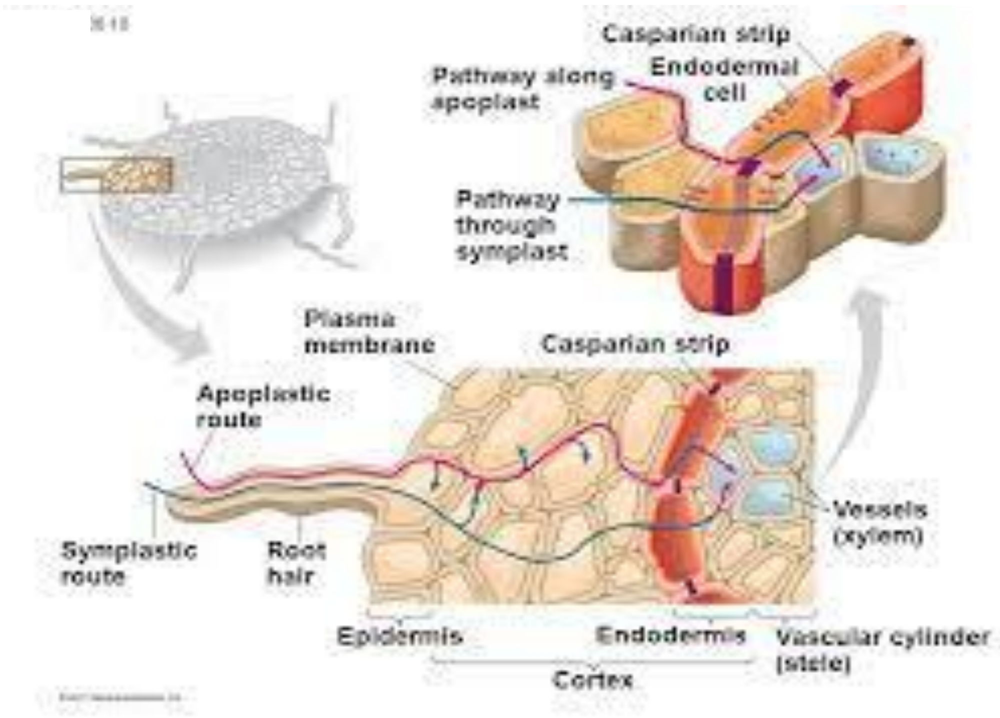

Fig. 4. Pathways of water and mineral uptake and transport in roots (Berardino, 2017)

I.2.1.2.1.2.2. Foliar pathways

Heavy metal uptake by foliar surfaces occurs through stomata, cuticular cracks, lenticels, ectodesmata and aqueous pores (Shahid et *al*., 2017). In fact, absorption of foliar-deposited heavy metals takes place mainly through ectodesmata, which are non-plasmatic channels positioned mainly between subsidiary cells and guard cells in the cuticular membrane or epidermal cell wall (Fig. 5). Moreover, the cuticle present above the guard cell is comparatively more permeable as compared to epidermal cells. Li et *al*. (2019) showed that particular matter (PM) adsorbed on plant leaves is mainly retained by trichomes and cuticular waxes, but some metals linked to particular matter can enter inside plant leaf tissues. Foliar uptake of metals is considered as a surface phenomenon (Zhang et *al*., 2019); however, the adaxial cuticular features are key in assisting high metal absorption via adaxial surfaces. Przybysz et *al*. (2020) studied the transfer of Cu and Ni-rich particles in birch, and suggested that particles may enter inside plant leaves through stomata. Fernández et *al*. (2017) proposed that particles could enter inside the leaf tissue via pores present on the leaf cuticle and inside stomata. Like root uptake, foliar uptake of heavy metals may also occur in a dose dependent manner. For example, Przybysz et *al*. (2020) reported linear relationship between Ni contents in the leaves and Ni contents in moderately and heavily polluted sites at the Kola Peninsula, Russia. Similarly, a linear relationship was reported between foliar applied As level and As uptake by the fronds (Zhang et *al*., 2019). Therefore, it is highly necessary to assess the risk for human health due to consumption of polluted plants after foliar uptake. However, there exist very rare data regarding health risks in kitchen gardens/farms near atmospheric contamination sources (Shahid et *al*., 2017).

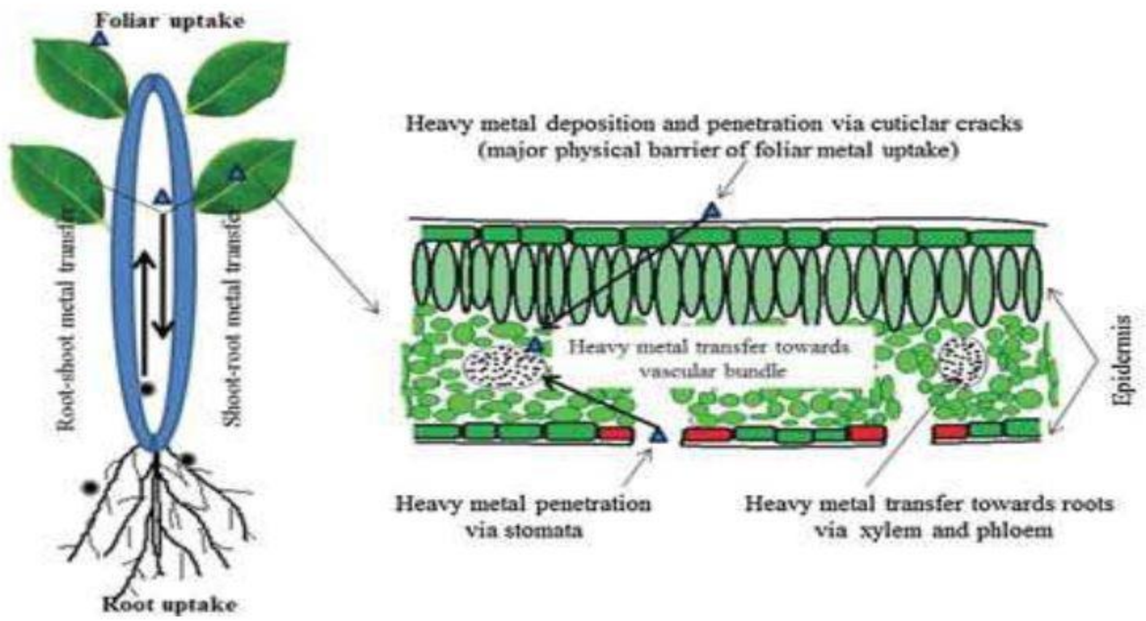

Fig. 5. Foliar pathways of heavy metal entrance to plants (Shahid et *al*., 2017).

I.2.1.2.1.3. Mobilities of heavy metals in the soil and their transfer to the aerial parts of the plant

I.2.1.2.1.3.1. Bioavailability of heavy metals

The amount of metal taken up by plants depends on its availability in the sediment, which is governed by a wide range of sediment and plant factors including pH, redox potential, salinity, cation exchange capacity, plant species and seasonal factors. Furthermore, the conditions existing around the root system may be very different from those in the bulk sediment (York et *al*., 2016; Bali et *al*., 2020). These physico-chemical parameters affected the (im)mobility of heavy metal as reported by (Palansooriya et *al*., 2020).

The mobility and bioavailability of metals are the factors which influence the chemical composition and sorption properties of soils (Sheoran et *al*., 2016; Devi & Bhattacharyya, 2018). For target heavy metals in soil, bioavailability is a critical factor affecting the efficiency of phytoextraction. To wetland macrophytes, heavy metal can have various bioavailabilities because of physiological differences with respect to uptake sites and uptake mechanisms (Jha et *al*., 2016). Soil pH is considered to be one of the most important chemical factors controlling the availability of heavy metals in soil. When metal is translocating in plants, the pH of soil increases also with the pH of soil solutions. According to Neina (2019), lower soil pH increases concentration of heavy metals in solution by promoting their desorption. Khan et *al*. (2018) reported that due to increase of pH, there was increase in the uptake of metals. That is the case of nickel where pH changes are related to Ni phytoremoval (Bernardi et *al*., 2020). Few fraction of soil metal can be uptake by plants and, for phytoextraction of lead, low bioavailability is considered as a major limiting factor (Mwilola et *al*., 2020).

Regarding the bioavailability of heavy metals/metalloids in soil, there can be three categories: readily bioavailable (Cd, Ni, Zn, As, Se, Cu); moderately bioavailable (Co, Mn, Fe) and least bioavailable (Pb, Cr, U) (Devi & Bhattacharyya, 2018; Rong et *al*., 2020). However, plants have developed certain mechanisms to solubilize heavy metals in soil. Plant roots secrete metal-mobilizing substances in the rhizosphere called phytosiderophores (York et *al*., 2016). Secretion of H^+ ions by roots can acidify the rhizosphere and increase metal dissolution. H^+ ions can displace heavy metal cations adsorbed to soil particles (Bali et *al*., 2020). Root exudates can lower the rhizosphere soil pH generally by one or two units over that in the bulk soil.

Furthermore, the rhizospheric microorganisms (mainly bacteria and mycorrhizal fungi) may significantly increase the bioavailability of heavy metals in soil (Ma et *al*., 2016; Da Silva et *al*., 2020). Interactions of microbial siderophores can increase labile metal pools and uptake by roots (Gkorezis et *al*., 2016; Kapahi & Sachdeva, 2019).

Phytoextraction of heavy metals can be practiced in two modes: natural and induced.

- In natural or continuous phytoextraction, plants are used for removal of heavy metals under natural conditions i.e., no soil amendment is made.
- In induced or chelate assisted phytoextraction, different chelating agents such as EDTA, citric acid, elemental sulfur, and ammonium sulfate are added to the soil to increase the bioavailability of heavy metals in soil for plant's uptake (Petruzzelli et *al*., 2015; Bali et *al*., 2020). The chelates form water-soluble complexes with the heavy metals in soil help in their desorption from soil particles.

Bioavailability of the heavy metals can also be increased by lowering the pH in soil since metal salts are soluble in acidic media rather than in basic media. However, these chemical treatments can cause secondary pollution problems. For example, synthetic chelate EDTA is non-biodegradable and can leach into groundwater supplies making an additional environmental hazard. *Helianthus annuus* (sunflower), in addition to a synthetic chelator, increase the bioavailability of metals such as Ni, Cd, Pb, Zn and Cu and showed a potential ability for phytoremediation of Ni contaminated soil (Devi & Bhattacharyya, 2018; Rong et *al*., 2020). Furthermore, synthetic chelating agents can also be toxic to plants at high concentrations. Thus proper care should be taken when practicing induced phytoextraction (Sheoran et *al*., 2016; Lajayer et *al*., 2019; Antoniadis et *al*., 2021). However, use of citric acid as a chelating agent could be promising because it has a natural origin and is easily biodegradable in soil. Furthermore, citric acid is not toxic to plants, therefore plant growth is not limited (Yu et *al*., 2019).

I.2.1.2.1.3.2. Transfer of some heavy metals

I.2.1.2.1.3.2.1. Cadmium (Cd)

Cadmium is one of the most toxic and mobile elements in the environment, which can cause serious health problems and can be lethal to plants at a low concentration of 2.5 mg/kg (Cai et *al*., 2019). It is considered the most serious pollutant of the modern age (Usman et *al*., 2019). Cadmium is highly mobile and it has been hypothesised that it could enter root cells through the same uptake processes that displace metal ions from essential micronutrients such as zinc (Ismael et *al*., 2019).

Cadmium may also reduce plant biomass, the numbers of flowers or fruits, chlorophyll content and the ability of plants to absorb essential nutrients (Khan et *al*., 2018; Coakley et *al*., 2019). It bioaccumulates in several organs and is classified as carcinogenic (Hajeb et *al*., 2014). As it has similar physico-chemical properties and high mobility in relation to essential micronutrients such as zinc, Cd is easily assimilated by plants (Coakley et *al*., 2019). Song et *al*. (2016) and Bali et *al*. (2020) reported that roots and leaves effectively absorb Cd. Huybrechts et *al*. (2019) mentioned that cadmium is not an essential element for plant metabolism and that it can be highly phytotoxic, leading to rapid death. It is knowned to disturb enzyme activities, inhibit DNA-mediated transformation in microorganisms, interfere with the symbiosis between microbes and plants, and to increase the susceptibility of plants to fungal invasion (Ojuederie & Babalola, 2017; Rizvi et *al*., 2020).

Cadmium can remain immobilized in wetland soils under anoxic conditions resulting from the accumulation of organic matter in the soil, the formation of metal sulfides and a generally almost neutral pH (Jacob et *al*., 2013). International threshold values of cadmium in soil are between 1.40 - 3 mg/kg (Anonymous 3, 2001). At this level in most cases, cadmium is metabolically absorbed and easily transported to the other parts of the plant. Kubier & Pichler (2017) indicated that pH and redox potential are the main hydrogeochemical parameters affecting Cd mobility and are probably related to Cd sorption on mineral surfaces and Cd release from carbonates and sulfides, such as pyrite. Non-residual heavy metal fractions are highly bound to cadmium and therefore make them mobile and potentially bioavailable for uptake by plants (Devi & Bhattacharyya, 2018). Cd concentrations above the threshold limit values have been found to be carcinogenic, mutagenic and teratogenic in a large number of animal species (Li et *al*., 2016; Rong et *al*., 2020). Cd has also been implicated as an endocrine disruptor (Matthiessen et *al*., 2018).

Agriculture and combustion emissions have been reported as the main anthropogenic sources of Cd through phosphate fertilizers, sewage sludge, landfills, traffic, industrial and mining waste to the environment (Bigalke et *al*., 2017). Depending on the origin of phosphate rocks, Cd in phosphate fertilizers can exceed 200 mg/kg P_2O_5 (Azzi et *al*., 2019). The presence and behaviour of Cd has been investigated with regards to agricultural aspects (Bigalke et *al*., 2017), bioavailability (Khan et *al*., 2018) and environmental remediation (Khan et *al*., 2017). Cd uptake in organisms is strongly influenced by Zn concentrations in the substrate (Jacob et *al*., 2013), as both metals compete for the same uptake mechanisms. Therefore, higher uptake and translocation in plants are observed when the Cd/Zn ratio is relatively high (Coakley et *al*., 2019; Palusińska et *al*., 2020; Palusińska et *al*., 2020).

I.2.1.2.1.3.2.2. Lead (Pb)

Lead is a non-essential element in metabolic processes and can become toxic or lethal to many organisms, even if absorbed in small amounts. Pb is considered very dangerous for plants, animals and especially microorganisms. The concentration of Pb ranges from 10 to 67 mg/kg for surface soils worldwide with the mean concentration of 32 mg/kg (Nazarpour et *al*., 2019; Kinuthia et *al*., 2020). Unlike some metals such as copper, zinc and manganese, which are essential for various physiological processes of plants, lead is a highly toxic metal pollutant that interferes with the plant metabolic processes (Pourrut et *al*., 2011). Higher Pb concentrations in plant tissues accelerate the production of reactive oxygen species (ROS) responsible for the deterioration of the lipid membrane, which then hinders chlorophyll, the photosynthetic process and suppresses overall plant growth (Jalmi et *al*., 2018; Xu et *al*., 2019). Plants uptake and accumulate lead in their roots and the lead content decreases as follows: roots> leaves> stems (Guo et *al*., 2018). Petelka et *al*. (2019) indicated that Pb content of plants grown in uncontaminated areas ranged from 0.05 and 3.0 mg.kg^{-1}. Amin et *al*. (2018) also reported that the Pb concentration ranged from 10 to 25 mg.kg^{-1}, and the maximum Pb accumulation was detected in the roots. Batista et *al*. (2017) showed that Pb caused phytotoxic effects, including chlorosis, necrosis, root and shoot stunting and reduce biomass production on vetiver (*Vetiveria zizanioides* L.), sunflower (*Helianthus annuus* L.), elephant ear (*Alocasia macrorrhiza*) and ʽembaúba' (*Cecropia sp*.).

Pb poisoning in children causes neurological damage resulting in a decrease of intelligence, short-term memory loss, learning disabilities and coordination problems (Mason et *al*., 2014). Pb has been found to be responsible for a number of diseases in humans, such as chronic neurological disorder, particularly in foetuses and children. This eventually leads to changes in behavioral and attitude with a progressive delay (Schupp et *al*., 2020). Lead from anthropogenic sources can reach the concentrations exceeding 10,000 µg/g and the main sources are lead mines, lead waste, cell batteries, lead solder and farms, houses painted with lead paint from buildings and other structures (Schupp et *al*., 2020). In addition, Pb pollution could be due to wastewater discharge from garages containing oil directly dumped in the marshes.

I.2.1.2.1.3.2.3. Copper (Cu)

Copper is an essential nutrient for plant growth in low concentrations. Copper can become highly phytotoxic at high concentrations. Lange et *al*. (2014) reported that Cu levels in various plants from unpolluted areas in different countries ranged from 2.1 and 8.4 mg.kg^{-1}. The risk of Cu toxicity in plant species is clearly demonstrated by Chiou & Hsu (2019) in the study on a copper phytotoxicity and its accumulation in the leaves of *Ipomoea aquatica*, *amaranthus sp*., *Brassica rapa* and *Glebionis coronaria*. This study showed that there are different tolerance ranges for plants, but the critical level of Cu toxicity is in the range of 20 - 30 mg/kg for most plants. Fumes of copper can cause metal fumes fever with flu-like symptoms of hair and skin discoloration in humans, although no dermatitis has been reported, it can also cause irritation of the upper respiratory tract, a metallic taste in the mouth and nausea (Taylor et *al*., 2020). The average concentrations of copper allowed by the WHO/FAO in soils is 100 mg/kg (Fosu-Mensah et *al*., 2018).

Copper does not decompose in the environment and, therefore, it can accumulate in plants and animals when it is found in the soils. In the areas surrounding copper disposals, plants do not have a high diversity of vegetation because very few species are able to grow on copper-rich soils. Because of its effects on plants, the metal poses a serious threat to agricultural production and, under conditions of favourable soil acidity and organic matter content, it can strongly affect the functioning of some agricultural lands. According to Ballabio et *al*. (2018), Cu in soil naturally ranges from 2 to 100 $mg.kg^{-1}$ (average: 25 $mg.kg^{-1}$), but it can increase to high soil pollution from agricultural activities. In mobile form, the availability of copper increases with lower pH and its concentration in the soil is closely related to the particle size composition and pH of soil.

Copper contributes to biological and physiological processes in plants, including photosynthesis, respiration, carbohydrate distribution, nitrogen and cell wall metabolism, seed production and disease resistance (Huang et *al*., 2020). Normally, the Cu concentration in plant tissues ranges from 5 to 30 $mg.kg^{-1}$ (Ballabio et *al*., 2018), and their excess or deficiency affect plant growth with drastic effects on plant biomass production and yield. Moreover, in the soil, when the concentration exceeds 60 to 125 $mg.kg^{-1}$, it also becomes toxic to tolerant plants, negatively influencing their biological and physiological processes (Napoli et *al*., 2019).

I.2.1.2.1.3.2.4. Zinc (Zn)

Zinc is a natural constituent of soil in the terrestrial ecosystem. It is an essential element for plant growth, which is actively absorbed by the roots as it plays an important role in plant structure and function (Palusińska et *al*., 2020). Zinc is also an element commonly found in the organism, tissues and it is essential in the processes of regulating metabolism, protein synthesis, insulin production and brain function. Although, compared to other trace elements, excess zinc inhibits the function of many proteins, disrupts calcium and iron management, which can cause anemia. Zinc is much less toxic to animals and humans (Chrzan, 2015). In plants and animals, Zn interacts with various elements such as cadmium and copper initiating various physiological process (De Oliveira, 2019; Adamczyk-szabela et *al*., 2020). Zinc levels are linked to sites located in urban centers and along transportation routes.

According to Adesuyi et *al*. (2018), the variation in the level of zinc concentration in wetland soils may show the impacts of pollution from anthropogenic rather than lithogenic activities. However, the permissible limit of Zn in soils is 300.00 mg/kg (Anonymous 3, 2001). Cu and Zn are both essential elements for plants, microorganisms, animals and humans. Chemical and physical soil factors (physiological properties of plants) have determined the link between soil and water pollution, and metal uptake by plants.

I.2.1.2.1.3.2.5. Chromium (Cr)

Chromium (III) is an essential trace element, but chromium (VI) is known for its toxicity and carcinogenic properties. Both forms of Cr vary significantly with respect to their bioavailability in soil, translocation and toxicity within plants (Shahid et *al*., 2017). The presence of Cr in soils could be due to the disposal of waste consisting of lead-chromium batteries, colored polythene bags, discarded plastics, resulting in discharge of chromium-containing effluents and empty paint containers (Amos-Tautua et *al*., 2014; Joutey et *al*., 2015). Due to its high solubility in water and soil, Cr (VI) is regarded as a hazardous ion that contaminates groundwater and can be transferred through the food chain (Joutey et *al*., 2015; Kumar et *al*., 2019).

Cr pollution in the lowland could come from the peeling of chrome- motorcycles (which were cleaned in the river). The mobility of chromium in wetlands depends on the sorption characteristics of the soil, including clay content, iron oxide content and the amount of organic matter present, and is transported by surface runoff to surface waters in either soluble or precipitated form (Lange et *al*., 2019; Sharma et *al*., 2020). The plant could tolerate up to 75 $mg.kg^{-1}$ Cr applied and beyond this, plant mortality may occur. The weathering of Cr containing rocks and leaching of soils discharge significant Cr contents into the aquatic environment (Ertani et *al*., 2017). Cr contents in soil ranges between 10 and 50 mg kg^{-1} under natural conditions, however, its concentration in agricultural soils can reach up to 350 mg kg^{-1} of the soil (Ertani et *al*., 2017). The United States Environmental Protection Agency (USEPA) has listed Cr among the 14 most dangerous substances that can cause serious health issues in living organisms (Anonymous 4, 2000). Cr can have both beneficial and harmful effects on human health depending on its uptake, exposure time and oxidation state.

The trivalent form of Cr (III) is an important nutrient for humans and according to the World Health Organization, its daily ideal intake is between 50 and 200 $\mu g.day^{-1}$ for the metabolism of carbohydrates, proteins and fatty acids. However, its excess in the body poses serious health concerns. Moreover, hexavalent Cr (VI) is 10 - 100 folds more harmful than Cr (III), which can cause allergies and skin problems (Sharma et *al*., 2020).

Cr does not have any known biological role in plant physiology (Reale et *al*., 2016). It is generally perceived that excessive Cr levels in plant tissues may provoke several morpho-physiological and biochemical processes in plants (Ud-Din et *al*., 2015; Kamran et *al*., 2017). Hence, Cr toxicity is reported to affect plant growth and impedes their essential metabolic processes (Kumari et *al*., 2016). Typically, Cr toxicity reduces plant growth by inducing ultrastructural modifications of the cell membrane and chloroplast, persuading chlorosis in the leaves, damaging root cells, reducing pigment content, disturbing water relations and mineral nutrition, affecting transpiration and nitrogen assimilation and by altering different enzymatic activities (Reale et *al*., 2016; Anjum et *al*., 2017). All these toxic effects of Cr might be due to the over production of reactive oxygen species (ROS), which ultimately disrupt the redox balance in plants (Anjum et *al*., 2017).

Chromium appears to have no essential role in plant metabolism, hence, there is no specific mechanism for its uptake in plants (Sharma et *al*., 2020). The distribution and translocation of Cr within plants depend upon the plant species, the oxidation state of the Cr ions, and also its concentration in the growth medium (Shahid et *al*., 2017). Compared to other heavy metals, the mobility of Cr in the plant roots is low. Therefore, the concentration of Cr in the roots is sometimes 100 times higher than in shoots (Sharma et *al*., 2020). For instance, Cr concentration was observed to be highest in the cytoplasm and intercellular spaces of rhizome and root cell walls of *Iris pseudacorus* (Caldelas et *al*., 2012). The higher accumulation of Cr in roots might be attributed to the sequestration of Cr in the vacuoles of root cells as a protective mechanism (Sharma et *al*., 2020). Thus, this mechanism provides some natural tolerance to plants towards Cr toxicity. Furthermore, the translocation of Cr from the roots to the aerial shoots is very limited and it depends on the chemical form of Cr inside the tissue (Shahid et *al*., 2017). In plant tissue, the Cr (VI) is converted to Cr (III) that has the tendency of binding to cell walls, which hinders the further transport of Cr within plant tissues (Sharma et *al*., 2020).

I.2.1.2.1.3.2.6. Nickel (Ni)

Nickel is an element that is present in the environment only at very low levels. It is the most abundant element (24th number on earth) with an approximate concentration of 0.008% (Genchi et *al*., 2020). It is essential in small doses, but it can be dangerous when the maximum tolerable quantities are exceeded (Nkrumah et *al*., 2019). Natural sources of the nickel included wind-blown dust, derived from the weathering of rocks and soils, volcanic emissions, bushfire and vegetation (Masindi & Muedi, 2018; Hanfi et *al*., 2020). Nickel has the ability to form an alloy with other metals, suppress corrosion and increase resistance to high temperatures (Klapper et *al*., 2017). Various industries have used nickel in stainless steel because of its corrosion-resistant properties and resistance to high temperature. Nickel is used in food industries as a catalyst (Usman et *al*., 2012). Agricultural soils are the most affected by nickel contamination in the terrestrial ecosystem (Nkrumah et *al*., 2019) and its toxicity impacts the metabolic pathway that inhibits photosynthesis (Genchi et *al*., 2020; Bernardi et *al*., 2020).

Accumulation of contaminants makes certain metabolic changes in an organism that tolerates heavy metal toxicity. Reactive oxygen species (ROS) and hydrogen peroxide substances are produced in response to stress on metal accumulation in the vacuole defense mechanism (Xu et *al*., 2019). High levels of nickel (Ni^{2+}) can pose a major threat to human health and the environment (Genchi et *al*., 2020). According to Anonymous 3 (2001), the permissible limit of nickel in soil is 50 mg/kg. The

International Agency for Research on Cancer (IARC) has included Ni and some of its compounds as probable human carcinogens. Arif et *al*. (2016) indicated that Ni is non-essential for the growth of healthy plants, animals and soil microbes. However, Bernardi et *al*. (2020) during their survey suggested that nickel was an essential element in many plants species and it interacts with iron found in haemoglobin and helps in oxygen transportation. Regarding plant metabolism, Ni is a key metal that stimulates plant enzyme systems. Ni is easily transported from roots to the tissues of the aerial plant parts.

I.2.1.2.1.3.2.7. Cobalt (Co)

Cobalt is not an essential element for plant growth, but it is considered beneficial for plants, animals and humans (Minz et *al*., 2018). Its concentration in plants is important because of its essential nature for animal nutrition. It is a component of vitamin B12, which all animals need. Although Co has beneficial effects on some plants, it is toxic at a higher level for many (Minz et *al*., 2018). Toxic concentrations vary widely in the range of 6 to 143 $mg.kg^{-1}$, depending on each plant specie (Akeel & Jahan, 2020). Moreover, if the soil is acidic, its potential for co-toxicity would be the greater (Hasan et *al*., 2011). Plant’s demand for cobalt is generally low. The concentration of cobalt in plants varies from about 8 to 100 $mg.kg^{-1}$ dry weight (Akeel & Jahan, 2020). Higher concentrations are toxic to plants, severely interfering with their metabolic functions (Sree et *al*., 2015). The average concentration of cobalt in plant tissues varies from 0.03 to 0.55 $mg.kg^{-1}$ of dry matter (Kosiorek, 2019).

The accumulation of cobalt in plant tissue causes irreparable damage to plant cells and is manifested on membranes as reduced growth and biomass, water and nutrient uptake, chlorosis and increased cell toxicity (Palansooriyaa et *al*., 2020; Akeel & Jahan, 2020). It has also been reported to suppress the synthesis of chlorophyll pigments by blocking the biosynthetic pathway (Vatansever & Ozyigit, 2017). It inhibits plant growth, photosynthesis and seed germination and may also cause neurotoxicity in animals and memory impairment in humans (Kosiorek, 2019; Palansooriyaa et *al*., 2020). Cobalt is also responsible for limiting root infections and initiating the process of nodule formation (Akeel & Jahan, 2020). Excessive concentrations of cobalt in plants can also contribute to retarding root and shoot growth, which in turn can disrupt water and nutrient uptake by plants.

I.2.1.2.1.3.2.8. Arsenic (As)

Arsenic is one of the toxic metal which has metallic and non-metallic properties, and it occurs in the following oxidation forms: elemental arsenic (As^0), arsine (As^{3-}) and arsenite (As^{3+}), arsenate (As^{5+}) (Pierre et *al*., 2018). It is generally ranked 12th among the elements in abundance in human body, 20th among the elements on the earth’s surface and 14th among the elements in seawater (Khalid et *al*., 2017). Arsenic is available in different chemical forms and can be converted by geochemical processes. Previous studies have reported that the inorganic form of arsenic such as, As (III) is 60 times more toxic than the organic arsenic As (V) (Chakraborty et *al*., 2014). Although As (III) is commonly rare, it can be found in groundwater, while organic As (V) is mainly present in toxic water which is a thermodynamic more stable form of arsenic (Al-Makishah et *al*., 2020). Usually, in unpolluted environment, the concentration of arsenic does not exceed 1 or 2 µg/L, while in the contaminated area, its concentration is reported to be 5000 µg/L, due to volcanic eruption and chemical weathering, which is 500 times higher than the permissible limit of arsenic concentration for drinking water (10 µg/l) (Anonymous 5, 2017). The bioaccumulation of arsenic in crop plants is supposed to enter in the human food chain pathways that cause numerous chronic diseases such as kidney and liver disorders, brain cancer, skin lesions, skin cancer, gangrene, black foot diseases and encephalopathy etc. (Rahman et *al*., 2018; Palansooriya et *al*., 2020).

Nowadays, numerous arsenic hyperaccumulators have been discovered including *Holcus lanatus* (Hartley-Whitaker et *al*., 2001) and *Vallisneria neotropicalis* (Lafabrie et *al*., 2011), etc. Additionally, several macrophytes have also been used for arsenic bioaccumulation (Chen et *al*., 2017). Sarkar & Paul (2016) and Khalid et *al*. (2017) reviewed the human-induced or anthropogenic sources of arsenic (pharmaceutical field, such as asphenamine-A($C_{12}H_{24}N_2As_2O_2$) and arsenic trioxide (As_2O_3), herbicides, some pesticides (35 g/kg) and fungicides, crop desiccants an wood preservatives), and they found that every year an excessive amount (l52,000 - 1120,000 tons) of arsenic was discharged into the environment. River water used for irrigation is also responsible for increased arsenic contamination due to untreated industrial effluents having arsenic toxicity. The arsenic content in soil varies according to the characteristics of parent materials available in the soil between 5 and 10 mg/kg. arsenic is usually present in the baseline soil (Basu et *al*., 2014).

I.2.1.3. Toxic effects of heavy metals and environmental risks

I.2.1.3.1. Impacts of heavy metals on the environment

Soils contaminated with metal trace elements may pose both direct threats (negative effects of metals on crop growth and yield) and indirect threats (entering the human food chain with a potentially negative impact on human health) (Masindi & Muedi, 2018). However, heavy metals are known to have impacts in soil ecosystems. Plants that grow in such environment mostly feel the impact of heavy metal contamination of soil. Some of these impacts include the decrease in seed germination and lipid content, decreased enzyme activity and plant growth, inhibition of photosynthesis, reduction of seed germination, reduction of chlorophyll production and plant growth (Minz et *al*., 2018; Haider et *al*., 2021). Recent studies reported that, excess Pb in plants can provoke the inhibition of the plant growth while Cd can inhibit photosynthesis and mineral assimilation causing leaf chlorosis, necrosis, and abscission (Adesuyi et *al*., 2015; Jha et *al*., 2016).

The presence of large amount of heavy metals in soil could also lead to the prevention of plants growth, uptake, physiological and metabolic processes, chlorosis, and harm to root tips, minimized water and uptake of nutrients and impairment to enzymes (Elmorsi et *al*., 2019; Elmorsi et *al*., 2019). Herbal plants can be used as a bioindicator of environmental changes for the reason that, they are sensitive to unfavorable soil conditions, especially in reference to heavy metal pollution. The trace elements detected in soil only reflect the information about the sampling location, but the metal uptake by plants gives information about their accumulative effects (Abedi & Mojiri, 2020; Huang et *al*., 2020). Furthermore, the potential detrimental effects of heavy metal polluted wastewater effluents on the quality of receiving water bodies are numerous, although it may depend on the volume and composition of the effluent that is discharged (Proshad et *al*., 2018; Yi et *al*., 2020). As an example, in aquatic ecosystems, the concentration and availability of lead can conduct to decrease dissolved oxygen, which may make young aquatic organisms, such as young fish vulnerable to lead than the adult fish and may provoke blackening of the tail region and spiral deformity to young fish (Madu et *al*., 2017; Ntsama et *al*., 2020).

I.2.1.3.2. Heavy metal toxicity and limited values in soil and plants

Toxicity of metals results not only from their presence in the environment, but primarily from their biochemical involvement in metabolic processes and mechanisms associated with absorption, accumulation and excretion carried out by living organisms. Capability of heavy metals to penetrate higher plants depends on soil properties and conditions prevailing in the environment, as well as on the physical and chemical form in which the element occurs (Fomenky et *al*., 2017; Akortia et *al*., 2019; Salem et *al*., 2020). The potential risk may be expressed with the accumulation index, representing the ratio between the average concentration of the element in the body and its content in soil (Mandeng et *al*., 2019; Obasi et *al*., 2019; Amaro-Espejo et *al*., 2020). Heavy metal species can have different bioavailability to wetland plants because of physiological differences with respect to uptake sites and uptake mechanisms (Jha et *al*., 2016).

Toxic metals can also be trapped once they are inside the cells. In this way, organic acids have been described to take part in heavy metal absorption, transfer and accumulation in plants (Amin et *al*., 2018). Furthermore, organic acids combined with heavy metals, reduced the combination opportunities of these metals with cellular proteins and enzymes, alleviating the damage caused by heavy metals (Shahid et *al*., 2015; Anjum et *al*., 2017; Shahid et *al*., 2017). Excess amount of heavy metal toxicity causes inhibition of nutrient uptake. Cell division plus their rapid growth, stop and disrupt photosynthesis thereby affecting plant development (Sheoran et *al*., 2016). A large number of factors regulates metal accumulation and bioavailability including pH, climatic conditions and plant genotype (Nedjma et *al*., 2019; Merhabi et *al*., 2019; Kumar et *al*., 2020; Mishra & Kumar, 2021).

Ni possess high rate of mobility in rhizospheric soils than in lead (Pb) and copper (Cu) (Nkrumah et *al*., 2019; Ekoa Bessa et *al*., 2021).

Toxicity of such metals is a resultant of their concentration (Table I), chemical form and features of the affected species (Shahid et *al*., 2017; Khalid et *al*., 2017) (Table II). The elements are accumulated mainly in topsoil, from which, via plants, they reach subsequent links of trophic chains, and simultaneously cause mutagenic and carcinogenic changes in organisms (Korish & Attia, 2020). Soils are increasingly threatened by heavy metal pollution due to their emission from motor vehicles, which imposes a specific band arrangement of the contaminated areas (Rodríguez-Eugenio et *al*., 2018; Merga et *al*., 2020). Heavy metal content in soils adjacent to roads is associated with traffic concentration, distance from the road, type of land and its use, as well as physical and chemical properties of soil and climatic conditions (Sheoran et *al*., 2016; Wang et *al*., 2019).

I.2.1.3.3. Effects of heavy metals and human health risks

Although heavy metals are present in soil as natural components, it is also a result of human activity. However, heavy metals even at low concentration can cause toxicity to human and other forms of life. Table III shows their adverse effects on human health (Kumar et *al*., 2017; Singh et *al*., 2018).

Table I. Examples of heavy metal contaminated soils exceeding permissible limits (Shahid et *al*., 2017; Khalid et *al*., 2017).

Heavy metal	Concentration in soil (mg/kg)	Maximum allowable limit	Fold-higher than allowable limit	Study area	References
Cd	42	3	14.0	Southern Italy	Baldantoni et *al*., 2016
	19		6.4	India	Tiwari et *al*., 2011
	16		5.4	Switzerland	Quezada-Hinojosa et *al*., 2015
	14		4.7	Mexico	Torres et *al*., 2012
	14		4.6	China	Shi et *al*., 2015
Pb	4500	100	45.0	China	Luo et *al*., 2011
	1988		19.9	China	Niu et *al*., 2015
	711		7.1	Uk	Nabulo et *al*., 2011
	452		4.5	Uganda	Nabulo et *al*., 2012
	302		3.0	Brasil	Carvalho et *al*., 2014
As	7490	20	374.5	Spain	Beesley et *al*., 2014
	4357		217.9	Italy	Marabottini et *al*., 2013
	354		17.7	China	Wei et *al*., 2015
	131		6.6	Korea	Myoung Soo Ko et *al*., 2015
	64		3.2	Bolivia	Acosta et *al*., 2015

Heavy metal	Concentration in soil (mg/kg)	Maximum allowable limit	Fold-higher than allowable limit	Study area	References
Zn	3833	300	12.8	China	Niu et *al*., 2015
	370		1.2	Nigeria	Obiora et *al*., 2016
	1168		3.9	Germany	Shaheen et *al*., 2014
	905		3.0	Portugal	Anjos et *al*., 2012
	393		1.3		Kwon et *al*., 2015
Ni	2603	50	52.1	Mexico	Torres et *al*., 2012
	373		7.5	Spain	Arenas-Lago et *al*., 2016
	201		4.0	Zimbabwe	Mapanda et *al*., 2007
	200		4.0	Turkey	Avci & Deveci, 2013
	153		3.1	China	Wang et *al*., 2015
Cu	35.582	100	355.8	Mexico	Torres et *al*., 2012
	19.581		195.8	Australia	Sacristăn et *al*., 2016
	448		4.5	China	Wang et *al*., 2015
	235		2.4	Portugal	Anjos et *al*., 2011
Cr	4309	100	43.1	Spain	Arenas-Lago et *al*., 2016
	590		5.9	China	Xu et *al*., 2014
	418		4.2	Greece	Panagopoulos et *al*., 2015
	224		2.2	Germany	Shaheen et *al*., 2014

[a] (Anonymous 6, 2007).

Table II. Accumulation of heavy metals in edible parts exceeding permissible limits of vegetables and crops (Khalid et *al*., 2017)

Heavy metals	Vegetables	Concentration in soil (mg/kg)	Concentration in plants edible parts (mg/kg)	[a] Maximum allowable limit	Fold-higher than allowable limit	References
Cd	*Lactuca sativa*	1.3	130	0.2	650	Pereira et *al*., 2011
	Solaum Lycopersicum	11.24	13		65	Hediji et *al*., 2015
	Agaricus bisporus	/	10		50	Schlecht and Säumel, 2015
	Cynosurus cristatus	0.2	9.0		45	Quezada-Hinojosa et *al*, 2015
	Brassica napus	1	6.0		30	Wu et *al*., 2012
Pb	*Daucus carota*	0.01	390	1	390	Carvalho et *al*., 2015
	Solanum aethiopicum	452	144		144	Nabulo et *al*., 2012
	Brassica oleracea	2.50	49		49	Perveen et *al*., 2012
	Lactuca sativa	/	28		28	Kang et *al*., 2013
	Spinacia oleracea	66.78	20		20	Khan et *al*., 2013
As	*Nicotina glauca*	14.660	92	0.15	613	Santos-Jallath et *al*., 2012
	Lactuca sativa	5.83	14		96	Caporale et *al*., 2014
	Oryza sativa	/	1.3		8	Smith et *al*., 2008
Zn	*Nicotina glauca*	507	1985	50	40	Santos-Jallath et *al*., 2012
	Brassica juncea	190	201		4.0	Mapanda et *al*., 2007
	Zea mays	80	148		3.0	Avci & Deveci, 2013
	Lactuca sativa	/	118		2.4	Bosiacki & Tyksinshi, 2009
	Spinacia oleracea	124	84		1.7	Naser et *al*., 2012

Heavy metals	Vegetables	Concentration in soil (mg/kg)	Concentration in plants edible parts (mg/kg)	[a] Maximum allowable limit	Fold-higher than allowable limit	References
Ni	*Lactuca sativa*	1.11	48	0.2	238	Perveen et *al*., 2012
	Solanum Lycopersicum	1.11	43		215	Perveen et *al*., 2012
	Portulaca oleracea	/	36		181	Renna et *al*., 2015
	Diplotaxis Tenuifolia	/	35		175	Renna et *al*., 2015
	Cupressus sempervirens	11.3	7.0		35	Farahat & Linderholm, 2015
Cu	*Solanum Lycopersicum*	/	202	10	20	Liu et *al*., 2006
	Coriandrum sativum	/	48		5	Gupta et *al*., 2013
	Zea mays	41	47		5	Avci & Deveci, 2013
	Agaricus bisporus	/	36		4	Liu et *al*., 2015
	Apium graveolens	46.85	11		1	Chao et *al*., 2007
Cr	*Solanum aethiopicum*	256	65	1	65	Nabulo et *al*., 2012
	Brassica oleracea	12.78	24		24	Tiwari et *al*., 2011
	Capsicum	1.11	17		17	Perveen et *al*., 2012
	Sinapis	1.11	13		13	Perveen et *al*., 2012
	Coriandrum sativum	1.11	13		13	Perveen et *al*., 2012
Mn	*Allium cepa*	573	585	500	1.17	Chiroma et *al*., 2014
	Lactuca sativa	619	512		1.02	Chiroma et *al*., 2014

[a] (Anonymous 6, 2007).

Table III. Types of heavy metals, permissible level, health hazards and sources health (Kumar et *al*., 2017; Singh et *al*., 2018).

Metal contaminant	EPA Regulatory Limit (ppm)	Health hazards	Majors sources	References
Chromium (Cr)	0.1	Allergic dermatitis, producing lung tumors, human carcinogens	Steel industry, mining, cement, paper, rubber, metal alloy paints	Salem et *al*. 2000; Cobbina et *al*., 2015;
Mercury (Hg)	2.0	Corrosive to skin, eyes and muscle membrane, dermatitis, nervous and kidney damage, anorexia, protoplasm poisoning, severe muscle pain, lung and kidney failure	Pesticides, batteries, paper and leather industry, thermometers, electronics, amalgam in dentistry, pharmaceuticals	Neustadt & Pieczenik, 2007; Gulati et *al*., 2010
Arsenic (As)	0.01	Bronchitis, carcinogenic dermatitis, liver tumors, gastrointestinal damage (GIT)	Pesticides, fungicides, metal smelters, coal fumes, wood preservatives	Tripathi et *al*., 2007; WHO 2010
Cadmium (Cd)	5.0	Kidney damage, bronchitis, carcinogenic, gastrointestinal disorder, bone marrow, cancer, weight loss	Welding, electroplating, pesticides, fertilizers, Cd-Ni batteries, nuclear fission plant	Degraeve, 1981;
Lead (Pb)	15	Mental retardation in children, liver, kidney, gastrointestinal damage (GIT), causes sterility, anemia, muscle and joint pains, hypertension	Paint, pesticides, smoking, batteries, water pipes, automobile emission, mining, burning of coal, lamps	Salem et *al*., 2000; Wuana & Okieimen, 2011; Padmavathiamma et *al*., 2007
Nickel (Ni)	0.2 (WHO permissible limit)	Causes Chronic bronchitis, reduced lung function, nasal sinus, cancer of lungs, affects fertility, hair loss	Steel industry, mining, magnetic industry	Salem et *al*., 2000; Khan et *al*., 2007; Duda et *al*., 2008
Copper (Cu)	1.3	Brain and kidney damage, long term exposure causes irritation of nose, mouth, eyes, headache, stomach ache, dizziness, diarrhea, chronic anemia	Brass manufacture, electronics, electrical pipes, additive for antifungal	Salem et *al*., 2000; Wuana & Okieimen, 2011
Zinc (Zn)	0.5	Nervous membrane and skin damage, causing short term illness called mental fume fever and restlessness, fatigue	Plumbing, refineries, brass manufacture, metal plating	Hess et *al*., 2002

I.2.2. Generalities on the lowland areas

I.2.2.1. Definition

Lowlands are primary landscape elements of the ecosystem, which present important planning and sanitation challenges. It is considered as wetland with the importance of flood control, groundwater recharge, sediment traps, atmospheric equilibrium and waste treatments (Anonymous 1, 2013; Adesuyi et *al*., 2018). According to Asibor (2009), area that possess water, plants and soils can be designated as wetland.

In Yaounde, the political capital of Cameroon, the uncontrollable speed of urbanization made people to be interested by lowlands and the near by peri-urban villages (Kimengsi et *al*., 2017). These swampy lowland areas are located at the bottom of valleys and normally unbuildable. Temple et *al*. (2009) reported that market gardeners used 55% of the swampy lowland areas for vegetable production, while 26% of food crops were grown on the slopes and 19% on the plateau are used for fruit and vegetable production. Lowland areas are essential for agricultural asset at the local and national levels and contribute to the ecological balance, biodiversity and the fragmentation of urban space creating a challenge for town planning. Lowland areas can contribute significantly to food production and poverty reduction in the sub-saharian region of Africa. Therefore, increasingly colonized by housing, agricultural and market gardening activities associated with breeding and fish farming, lowlands are polluted and heavy metals are the worst environmental contaminants (Briffa et *al*., 2020).

I.2.2.2. Characteristics of lowlands in Cameroon

As an ecosystem, lowlands are characterized by the presence of plants that are adapted to live in the soil formed under saturated conditions. A typical characteristic of flooded wetland soil is the reduction of oxygen leading to anoxia in these areas. This results in the accumulation of organic matter in the soil, the formation of metal sulfides, and the pH tends towards neutrality. Cadmium can remain immobilized in wetland soils under these conditions (Haider et *al*., 2021). Wetland soils are also known to show different biogeochemical behavior when compared to dry land soils.

Plants growing in flood-prone lowland areas are exposed to anthropogenic sources of pollution like landfill leachates, fertilizer erosion from agricultural run-offs, herbicides and pesticides, industrial effluents, vehicle emissions, anarchic spill of used hydrocarbons and oils from garages. In addition, other sources such as fertilizer erosion from agricultural run-offs, sewage from household and hospital effluents, urban runoffs, incineration of solid wastes and household ashes, irregular disposal of batteries, paints and metal plating coming from human activities (Garba & Abubakar, 2018; Zwolak & Sarzy, 2019; Yang et *al*., 2020; Zhou et *al*., 2020).

I.2.2.3. Mineralogy and geology of the Yaounde soils (Cameroon)

Soil is the key component of geo-ecosystems, characterized by specific physical, chemical and biological properties developed under the impact of soil formation proceses over centuries, as well as agricultural and non-agricultural human activities. Cameroon is constituted of 75 to 100% of acid soils and its surface is covered by 21.7 million hectares by humid forest zones (Tandzi et *al*., 2018). A geological survey carried out by Ngon Ngon et *al*. (2009) in Yaounde has revealed the presence of homogeneous clayey laterite in the upper part of a laterite cover on interfluves, thickest on hills (780–800 m altitude) where ferricrete is absent, and heterogeneous hydromorphic clayey material present in valleys. The sediments are mainly composed of quartz, kaolinite, accessory goethite, smectite, rutile, feldspars, illite, gibbsite, and interstratified illite-vermiculite (Ekoa et *al*., 2018). These soils are characterized by an excess of Al^{3+}, Mn^{2+} and H^+ with deficiencies in Ca^{2+}, Mg^{2+} and PO_4^{3+}, reducing root growth of plants and the absorption of the essential nutrients (Krstic et *al*., 2012). At Simbock lake in Yaounde, the sediments have low contents in Al_2O_3, Fe_2O_3, Na_2O, K_2O, MgO, and CaO as well as high values in SiO_2, P_2O_5, TiO_2, and MnO relative to the upper continental crust (Ekoa et *al*., 2018). The clay raw materials are mostly made up of fine particles (ranging from 55 to 60% clay + silt in the clayey laterite, more than 70% clay + silt in the clayey hydromorphic material). Their chemical composition is characterized by silica (< 60% SiO_2), alumina (< 35% Al_2O_3) and iron (ranging from 3 to 14% Fe_2O_3). Their main clay minerals are disorganized and poorly crystallized kaolinites (Ngon Ngon et *al*., 2009). The study carried out by Ekoa et *al*. (2018) on the mineralogy and geochemistry of sediments from Simbock Lake,Yaounde area (southern Cameroon): provenance and environmental implications indicated that, the sediments are sandy, sand-clayey to clayey and yellowish brown to greenish brown, and with high amounts of organic matter (average value of TOC is 1.95%). At Nkolbisson in Yaounde, the soil texture is clay loam with a strong tendency for water logging, manganese toxicity and belongs to the Kandiudox type (Tekeu et *al*., 2015).

I.2.2.3.1. Physical properties of the lowlands in Yaounde (Cameroon)

In the humid forests zones of Cameroon, acidity is the main limiting factor of soil productivity. Ngonkeu (2009) showed that the use of improved maize varieties coupled to biological processes in soil fertility of certain species of arbuscular mycorrhizal fungi has the capacity to improve the tolerance of plants to acid soils with aluminum and manganese toxicity.

From a morphological point of view, the clays are very heterogeneous with several clay loam or sandy textures as cited by diverse studies (Ngon Ngon et *al*., 2009; Fokom et *al*., 2012; Kowalska et *al*., 2021). The large swampy valleys are at risk of water logging, which could inhibit plant growth by reducing the availability of oxygen and nutrients to the roots. There is also the potential leaching of nutrients into waterways through run-offs from rainfall and leaching if not taken up by plant roots.

According to the International Soil Classification System, the majority of soils in Cameroon are Ferralitic: yellow, ochre or red in color depending on the mother rock and the landscape and arise from migmatitic gneiss soil parent material (Ngon Ngon et *al*., 2009; Van Ranst et *al*., 2019). Soils of this type are categorized as clay and acidic with a pH in the range from 4 to 5.5 in $CaCl_2$ (Takoutsing et *al*., 2015).

I.2.2.3.2. Toxicity of soils in Cameroon

In Cameroon, acid soils cover 75% of arable land and it is defined as soils with pH < 5.5 in the top layer (Dalovic et *al*., 2012). Acid soil toxicity is caused by a combination of high solubility of toxic heavy metal elements (Fe, Cu, Mn, Zn and Al), a lack of essential nutrients (P, Mg, Ca, K, Na), and low soil pH (Tandzi et *al*., 2018; Neina, 2019). Low soil pH can therefore generate excesses of aluminum, iron and manganese, which hamper crop production (Tandzi et *al*., 2018). High Al and Fe oxides and hydroxide in low soil pH are responsible for phosphate fixation, making it unavailable to plants (Fink et *al*., 2016). All of these toxicities (Al, Mn, and Fe) should be considered when working in soil contaminated with heavy metal. Aluminum, iron and manganese toxicities are the main type and natural form of metal toxicity in Yaounde-Cameroon.

I.2.2.3.2.1. Aluminum toxicity

Under aluminiun (Al) toxicity, nitrogen (N), phosphorous (P), and potassium (K) uptake, which are essential nutrients responsible for the stimulation of root growth (Bojórquez-Quintal et *al*., 2017; Rahman et *al*., 2018), become unavailable. Phosphate deficiency leads to stunted plant growth, and thin and spindly stems with purpling leaves, which results in the reduction of grain yield (Temegne et *al*., 2018). Strong subsoil Al toxicity reduces plant-rooting depth, increases susceptibility to drought and decreases the use of subsoil nutrients (Rahman et *al*., 2018). Al toxicity effects result in root damage, which hamper nutrient uptake ability, resulting in nutrient deficiency in the plant (Tandzi et *al*., 2018).

Determination of the content of available Al (exchangeable and in the soil solution) is essential for an evaluation of the risk for plant production in acid soils. While most of the attention on acidic soils has been focused on Al toxicity, limited attention has been placed on Fe and Mn toxicities.

I.2.2.3.2.2. Iron Toxicity

Iron is the fourth most abundant mineral in the earth's crust after oxygen (O_2), silicon (Si), and (Al). Fe toxicity is a disorder associated with large concentrations of reduced iron (Fe^{2+}) in the soil solution, which occurs in flooded soils (Fink et *al*., 2016). The hydrolysis of Fe is more acidic than Al hydrolysis. Acidity resulting from Fe toxicity is normally buffered by Al hydrolysis reactions. However, once most of the soil Al ions have reacted, Fe hydrolysis takes over, leading to a profound decrease in soil pH (Tandzi et *al*., 2018).

In low soil pH, the anaerobic bacteria provide very high amounts of ferrous ion, which becomes toxic to plants. Acid soils that are poorly aerated or compacted can increase iron content to the point of toxicity. A high concentration of Fe^{2+} in the rhizosphere has antagonistic effects on the uptake of essential nutrients (P, K, and Zn) by the plants, causing the accumulation of harmful organic acids or hydrogen sulphides, and consequently leading to plant yield reduction (Rai et *al*., 2021). Yield reductions of 12 to 100% have been previously observed in rice growing in iron toxic soils (Sikirou et *al*., 2016), depending on the level of iron toxicity, genetic background of genotypes, and soil fertility status. High iron availability in the soils can also lead to direct or indirect toxicity in the plants (Saaltink et *al*., 2017). High toxic levels of accumulated Fe in plants can damage lipids, proteins, and deoxyribonucleic acid (DNA). Direct effects of iron toxicity also include damage to cell structures leading to reduced plant growth and injury to foliage (Saaltink et *al*., 2017).

I.2.2.3.2.3. Manganese Toxicity

Manganese (Mn) is an essential trace element throughout all stages of plant development, which becomes toxic when taken up in excessive quantities. Mn is deficient in plants when its level is less than 15 ppm and excessive or toxic when its concentration is higher than 200 ppm (Tandzi et *al*., 2018). Despite the importance of Mn for photosynthesis and other processes, the physiological relevance of Mn uptake and compartmentation in plants has been underrated (Alejandro et *al*., 2020) Mn toxicity is associated with Al and Fe hydrolysis, the primary reactions causing soil acidity. Soil acidification further enhances the solubility of Mn, and thus increases its bioavailability to toxic levels in natural and agricultural systems (Zaitsev et *al*., 2020). However, Mn deficiency is a serious, widespread plant nutritional disorder in dry, well-aerated and calcareous soils, as well as in soils containing high amounts of organic matter, where bio-availability of Mn can decrease far below the level that is required for normal plant growth. By contrast, Mn toxicity occurs on poorly drained and acidic soils in which high amounts of Mn are rendered available (Alejandro et *al*., 2020). The effects of Mn toxicity are more pronounced in sensitive plants with a decrease in soil pH, which further increases the solubility of Mn (Blamey et *al*., 2018). The first symptoms of Mn toxicity appear on the oldest leaves of plants as chlorosis, which later progresses to necrosis (Zaitsev et *al*., 2020). In addition, plants exposed to excess Mn exhibit a very strong inhibition of chloroplast structure and functions, reduced photosynthetic and transpiration rates, and inhibition of carbon dioxide (CO_2) fixation as a result of stomatal closure (Alejandro et *al*., 2020). To date, there is a very limited number of published reports on manganese toxicity in plants. Therefore, this area of study requires more investigations.

I.2.2.4. Importance of lowlands

As an ecosystem, lowlands are valued for their contribution to ecological balance and biodiversity. They also aid in food production especially in the cultivation of rice and vegetables. Rapid urbanization and industrialization have led to increase in pollution from landfill leachates, industrial effluents, vehicle emissions, fossil fuels, fertilizer erosion from agricultural run-offs, herbicides and pesticides, sewage and municipal wastes. All these contributed to the accumulation of pollutants in nearby aquatic systems (Adesuyi et *al*., 2015a; Adesuyi et *al*., 2016). Among the worst environmental contaminants are the heavy metals (Briffa et *al*., 2020).

Lowland soils are also known to show different biogeochemical behavior when compared with dry land soils. Lowlands are very important subsystems of the general ecosystem as they play vital roles in the sustenance of both surface and ground water resources of the earth. The importance of any wetland is cited within its functions and values. Notably, the functions of wetlands include flood control, groundwater recharge, coastal protection, sediment traps, atmospheric equilibrium and waste treatments, as well as providing nurseries for aquatic life and habitat for upland mammals (Anonymous 1, 2013; Adesuyi et *al*., 2018). Eutrophication may inhibit macrophyte growth and consequently result in wetland stress (Kim et *al*., 2021). In most eutrophic ecosystems, phosphorus (P) is often the limiting factor.

I.2.3. Criteria for selecting phytoremediating plants in lowlands

I.2.3.1. Characteristics of plants selected for heavy metal phytoremediation

Plants suitable for phytoremediation should have the following characteristics (Khalid et *al*., 2017; Shah & Daverey, 2020):

- hyperaccumulation and hypertolerance;
- high growth rate;
- high translocation factors (TF) and high bio-concentration factor (BCF);
- production of above ground biomass;
- widely distributed and highly branched root system;
- translocation of the accumulated heavy metals from root to shoots;
- good adaptation to prevailing environmental and climatic conditions;
- resistance to pathogen and pest;
- easy to cultivation and harvest;
- repulsion to herbivores to avoid food chain contamination.

Desirable characteristics for efficient rhizofiltration include plant tolerance to metal concentrations, the ability to accumulate high concentrations of these elements, high biomass production and limited translocation of contaminants from roots to shoots (Rezania et *al*., 2016; Galal et *al*., 2017). A plant with high translocation of metals from roots to shoots reduces the benefits of the rhizofiltration process, as it increases the number of plant parts that are contaminated with metals (Emurotu & Onianwa, 2017; Liu et *al*., 2020; Dinu et *al*., 2020) and consequently, the risk of contamination of other organisms through the food chain.

Macrophytes that are developed in the lowland areas have the ability to grow fast, accumulate heavy metals and metalloids in large quantities, survive under harsh conditions and tolerate high concentrations of toxic elements (Adesuyi et *al*., 2018). In aquatic ecosystems, macrophytes have an important role. Njuguna et *al*. (2017) reported that macrophytes could remove, transform or stabilize heavy metals in water and sediments. Abedi & Mojiri (2020) complete that through roots or leaves macrophytes can uptake heavy metals either from sediment or from water. A good biomonitor will indicate the presence of the contaminant and also provide additional information about the quantity and intensity of the exposure (Alexandrino et *al*., 2020). Metal uptake by plants can be element specific, plant species specific and plant tissue specific (De Oliveira, 2019; Abedi & Mojiri, 2020).

I.2.3.2. Macrophytes diversity in lowlands

Macrophytes are considered as aquatic plants, growing in or near water that are either emergent, submerged or floating. They also produce oxygen, which helps in overall lake functioning, and provide food for some fish and other wildlife (Schneider et *al*., 2018). Plants growing in soils polluted with metals are referred to as metallophytes and they developed growing techniques:

- excluder's, which prevent a broad range of soil metal concentrations from entering their aerial parts;
- indicators, which take up metals at a linear rate relative to soil concentrations; and
- accumulators, which withstand the uptake of even higher metal levels than those of the soil.

Hyperaccumulators can accumulate over 1000 $mg.g^{-1}$ heavy metals in aboveground tissues (Chandra et *al*., 2018). A prerequisite is to tolerate high concentrations of metals efficiently within plant tissues and cells.

A study done by Njuguna et *al*. (2017) on the assessment of macrophyte, heavy metal, and nutrient concentrations in the water of the Nairobi River in Kenya shows that 31 plant species belonging to 23 families were prevalent and 11 species were identified as bioaccumulators. For instance, *Leersia hexandra* and *Pennisetum purpureum* have been found to effectively remove Cd and Zn, Cu, Mn in soils (Liu et *al*., 2015; Hassan et *al*., 2020). *Colocasia spp*. and *Amaranthus spp*. were good at extracting Cd from soil, while *Tithonia diversifolia* was effective in uptaking Pb (Njuguna et *al*., 2017). *Eichhornia crassipes* was found to efficiently remove NO_3^-, Fe, Zn, Cu, Cd, and Cr while *Cyperus rotundus* uptake Cd and Cr from contaminated water and soil, respectively (Yadav et *al*., 2015). *Ricinus communis* was found to accumulate Cd, Pb, Ni, As, and Cu from contaminated soil (Yeboah et *al*., 2020). *Cyperus articulatus* removed As, Cd, Cr, Cu, Fe, Hg, Mn, Ni, and Pb from water while *Typha domingensis* decontaminated Hg and Cr from water (Gomes et *al*., 2014; Sultana et *al*., 2014).

Accumulators and specifically hyperaccumulators have attracted considerable interest in recent years, particularly as potential remedies for heavy-metal contaminated soils and waters. Suman et *al*. (2018) further pointed out that many hyperaccumulators of heavy metals such as Cu, Co, As, Zn, Pb, Mn, Se, Cd, and Ni are characterized by their ability to survive high concentrations of these heavy metals and always endemic to metal-rich substrates. Over 420 species of heavy metal hyperaccumulators that belong to about 45 plant families have been identified (Adesuyi et *al*., 2018; Eid et *al*., 2020). Anum et *al*. (2019) reported that over 50 species of Ni hyperaccumulators have been found endemic to the metal-rich serpentine outcrops of New Caledonia (an island). These include species from genera such as Cledion (Euphorbiaceae), Argophyllum (Grossulariaceae), Casearia (Flacourtiaceae), Geissois (Cunoniaceae), Homalium (Flacourtiaceae), Hybanthus (Violaceae), Oncotheca (Oncothecaceae), Pancheria (Cunoniaceae), Phyllanthus (Euphorbiaceae), and Xylosma (Flacourtiaceae). Plants Cu tolerance varies with species and cultivar, but in general, plants are Cu-excluders and Cu-accumulators plants are uncommon. Napoli et *al*. (2019) reported that about 34 species have been discovered to be hyperaccumulators for Cu, among which *Ipomea alpine*, *Aeolanthus biformifolius*, *Eleocharis acicularis*, *Haumanias trumkatangense, Commelina communis*, *Rumex acetosa* and *Artemisia argyi,* but they produce few biomass and have slow growth.

Plants enable the direct assessment of the response of wetland vegetation to changes in aquatic discharge. In the past few decades, an increasing use of higher plant leaves as biomonitors for heavy metal pollution has been on the increase especially in fragile and urban areas (Alexandrino et *al*., 2020). Some of these plants include *Alternanthera philoxeroides*, *Commelina benghalensis*, *Eichhornia crassipes*, *Enhydra fluctuans*, *Ipomoea aquatica, Pennisetum purpureum*, *Ludwigia adscendens*, *Sagittaria sagittifolia*, *Pistia stratiotes* and the sedges.

Several fascinating patterns have been observed since the events of hyperaccumulators discovery (Khalid et *al*., 2017; Méndez et *al*., 2018). First, many families of plants such as Asteraceae, Brassicaceae, Euphorbiaceae, Fabaceae, Flacourtiaceae, and Violaceae were found to contain large numbers of hyperaccumulators. This implied that several genera within the families might be predisposed or preadapted to tolerate high level of heavy metals. The second pattern observed was the high percentage of these hyperaccumulators in tropical regions of the world. About 320 species of Ni hyperaccumulators discovered, two-third was found in the tropical regions of the world (El-amier et *al*., 2018; Anum et *al*., 2019). The third pattern indicated that more than 80% of the identified hyperaccumulators take up more Ni than the other metals.

Aquatic plants are known to accumulate metals from their environment. The aquatic plants in metallic pollution acted as biological filters and biomonitors of environmental metal levels (Rezania et *al*., 2016; Galal et *al*., 2017). Plant's process of metal removal by binding in soils, precipitation as insoluble salts is described by Awa & Hadibarata (2020), who also reported a model for the treatment of industrial effluents, municipal wastewater and eco-sustainable utilization of biomass using macrophytes.

I.2.4. Bioaccumulation capacities of plants and phytoremediation strategies

I.2.4.1. Plant bioaccumulation capacities

Plants grown on metal enriched soils take up metal ions in varying degrees. Generally, to quantify the bioaccumulation ability of the contaminants in the environment, two approaches are used with the assumption that plants achieve a chemical equilibrium with respect to a particular media or route of exposure (Galal et *al*., 2017; Kandziora-Ciupa et *al*., 2017; Liu et *al*., 2020). This approach used bioconcentration factor (BCF), bioaccumulation factor (BAF) and also the indice-like bioaccumulation coefficient (BAC) to estimate chemical residues in biota from measured concentrations in the appropriate reference media. Accumulation of selected metals varied greatly among plant species. The normal level in shoots of plants for Pb and Cu as given by (Amin et *al*., 2018; Lange et *al*., 2019) are 5 and 10 mg/kg. Cu concentration > 40 mg/kg of dry matter could induce toxicity in plants and cause toxic effects in animal feeding (i.e. sheep).

Bioconcentration factor is indicative of the degree of enrichment of a heavy metal in an organism relative to that in its habitat. It has been reported that bioconcentration factor (BCF) values of seven typical heavy metals in crop grains decreased exponentially with average concentrations of the metal in soil (Wang et *al*., 2017). Bioconcentration factor indicates the efficiency of a plant in up-taking heavy metals from soil and accumulating them into its tissues and can be used to assess a plant's potential for phytoremediation purposes. The bioconcentration factor (BCF) provides an index of the ability of the plant to accumulate the metal with respect to the metal concentration in the substrate. Coakley et *al*. (2019) mentioned that metal accumulations by macrophytes could be affected by metal concentrations in water and sediments. In the same light, Rezania et *al*. (2015) recorded that water hyacinth effectively removed appreciable quantity of heavy metals (Cd, Co, Cr, Cu, Mn, Ni, Pb and Zn) from freshwater especially at low concentrations. Coakley et *al*. (2019) found that the BCF values of Zn in water hyacinth roots and shoots decreased when the ambient water concentration of Zn increased. Similarly, Huang et *al*. (2020) determined that when the external environment had a low concentration of Cu level at 0.18 mg/l, the BCF of roots was highest at 6,166. Rezania et *al*. (2015) mentioned that high metal concentration is toxic to the growth of water hyacinth plant. Therefore, the bioaccumulation factor will increase with a low metal concentration and decrease with an increase in metal concentration.

I.2.4.1.1. Potential of bioaccumulation in plants

Plant species, physiological stage, uptake capability as well as growth rate are major determinants of metal transfer from the soil to the crop. Heavy metal uptake by plants has two patterns:

- shoot exclusion, where metals are accumulated in the root but translocation to the shoot is restricted; and
- accumulation, where metals are concentrated in the aerial parts (Dinu et *al*., 2020).

The ability of plants to tolerate and accumulate these metals may provide the bases for their phytoremediation usefulness.

Bioaccumulation factor (BAF) and translocation factor (TF) are used to assess the translocations of heavy metals into the growing plant tissues. The bioaccumulation factors for shoots (BAF) and the transfer factor (TF) have also been provided to understand their accumulation potential. Bioaccumulation factor (BAF) is used to quantify the toxic element accumulation efficiency in plants by comparing the concentration in the plant part and an external medium (Ali et *al*., 2019). BAF has been categorised as: <1 excluder, 1 - 10 accumulator and >10 hyperaccumulator (Jha et *al*., 2016). A number of workers has reported bioaccumulation of substances, including heavy metals. Parihar et *al*. (2020) reported in the study of bioaccumulation potential of indigenous plants in rural areas of, Punjab (India) that, high bioaccumulation of individual metals was observed in herbs like *C. sativa*, *M. polymorpha*, and *Amaranthus spp.*, and cumulatively, trees appeared to be the better bioaccumulators of heavy metals. In addition, various plants produce different enzymes that render to detoxify the heavy metal, and hence, enhance plant tolerance ability to resist against heavy metals. Different plants possess different capabilities for the same metal uptake. Green leafy vegetables have natural ability to absorb a huge concentration of multiple metals for their metabolic processes and growth as compared to non-green leafy vegetables (Iqbal et *al*., 2020). Two processes primarily control the movement of metal-containing sap from the root to the shoot, termed translocation: root pressure and leaf transpiration (Fig. 6). Some metals are accumulated in roots, probably due to some physiological barriers against metal transport to the aerial parts, while others are easily transported in plants (Coakley et *al*., 2019). Ni was found to be immobile in *Phragmites communis*, where it showed a concentration in roots 48 fold higher than in leaves (Bernardi et *al*., 2020). A plant’s ability to translocate metals from the roots to the shoots is measured using the translocation factor (TF). TF greater than 1 (TF>1) signifies that the plant effectively translocates heavy metals from roots to the shoots (Anum et *al*., 2019).

High accumulation of heavy metals in roots and low translocation in shoots may indicate appropriateness of a plant species for phytostabilisation (Radziemska et *al*., 2017). This mechanism of partitioning is a common strategy of plants to concentrate harmful ions in the roots in order to prevent toxicity to the leaves, which is the site of photosynthesis and other metabolic activities (Jha et *al*., 2016). Plant species with high TF values were considered suitable for phytoextraction and this requires the translocation of heavy metals easily in harvestable plant parts i.e. shoots (Dinu et *al*., 2020). According to Lajayer et *al*. (2019), phytoextraction is a process of soil decontamination without destroying soil structure and fertility.

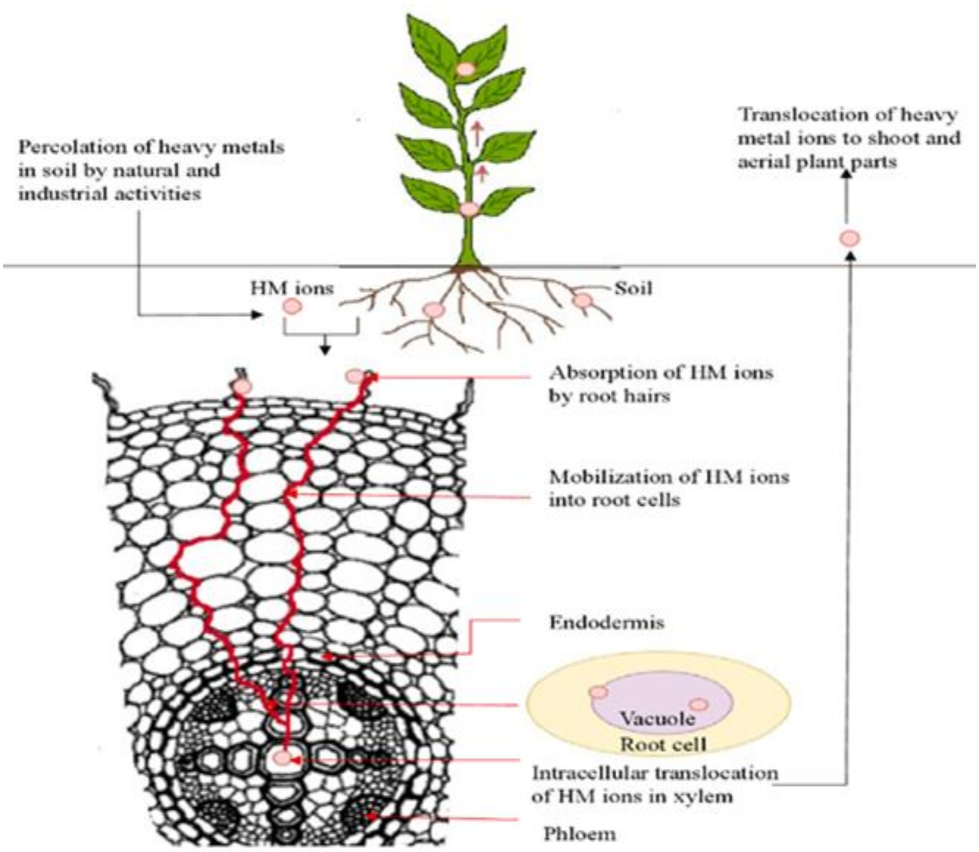

Fig. 6. Uptake, translocation and accumulation of heavy metal (HM) in plants (Dhalaria et *al*., 2020)

I.2.4.1.2. Factors influencing the accumulation capacity

The capacity of plants to hyperaccumulate trace elements is influenced by the presence of humic substances or other chelating substances, temperature and salinity (Wang et *al*., 2019; Grochowska, 2020). The persistence of heavy metal in the environment made them to enter to organisms and accumulate therein. As earlier mentioned, bioaccumulation of heavy metals in biota depends on some factors that influence their uptake. For instance, the uptake of heavy metals in plants depends on bioavailability of the metal in soil, which in turn depends on several factors such as metal speciation, pH and organic matter contents in soil (Ali et *al*., 2019). The amount of metal taken up by plants depends on its availability in the sediment, which is governed by a wide range of sediment and plant factors including pH, redox potential, cation exchange capacity, plant species and seasonal factors. Furthermore, the conditions existing around the root system may be very different from those in the bulk sediment (Correa-García et *al*., 2018; Benizri & Kidd, 2019). Metals, which are more bioavailable in soil, may be accumulated in plants more easily and thus will have more bioaccumulation potential. An assessment of bioaccumulation of heavy metals in plants may be used for an estimation of bioavailability of the metals in soil. Accumulation factors and translocation factors are measures used to identify and define hyperaccumulators. Accumulation factor is the ratio of metals in the shoot tissue to the metal in the contaminated environment. Translocation factor is defined as the ratio of metals in the shoot to the metals in the root of the plants. Furthermore, accumulation factor is also necessary for identifying the feasibility of phytoextraction. This feasibility of phytoextraction means the number of cropping cycles required for the metal removals to the level that is accepted (Yu et *al*., 2019).

Based on accumulation factors, translocation factors, high biomass and fast growing rate, over 400 species of hyperaccumulators that belong to 45 families have been identified as effective hyperaccumulators of metals from contaminated soils with species specifically capable of phytoextraction in mine tailings (Anum et *al*., 2019; Dinu et *al*., 2020).

I.2.4.2. Phytoremediation strategies

I.2.4.2.1. Definition

Phytoremediation comes from the Greek word phyto, meaning plant, and the word remedium, in Latin, meaning balance or remediation. Phytoremediation is defined as the use of green plants to reduce the concentration or toxic effects of contaminants in the environment (Yan et *al*., 2020). It is a technique that could potentially help for the removal of heavy metals and radionuclides as well as organic pollutants (polynuclear aromatic hydrocarbons (PAHs), polychlorinated biphenyls) and pesticides. According to Singh et *al*. (2017), phytoremediation is a method used to clean-up environmental pollutants by plants. Generally, water, air and soil are the constituent elements of the environment and the process of removing any contaminant or pollutant from the environment is called environmental remediation.

Due to the fact that, heavy metal continuously increase in soil ecosystems during the years (Suman et *al*., 2018; Ashraf et *al*., 2019; Huang et *al*., 2020), soils clean-up techniques have been applied for the maintenance of the environment health and ecological restoration. It was categorized into physical, chemical and biological methods (Hasegawa et *al*., 2016; Sarwar et *al*., 2017; Razzaq, 2017). Traditionally, remediation of metals contaminated soils includes techniques such as precipitation, reverse osmosis, evaporation, ion exchange, chemical reduction, soil incineration, excavation, soil washing, soil flushing, solidification and stabilization of electro-kinetic systems (Sharma et *al*., 2018; Upadhyay et *al*., 2019). Thus, it is recognized that these on-site management processes have their limitations. According to Ali et *al*. (2013); Suman et *al*. (2018); Upadhyay et *al*. (2019), they are costly, have irreversible changes in soil properties, disturbance of native soil microflora and required much resources. Theses authors presented that the cost of cleaning up one acre of sandy loam soil with a contamination depth of 50 cm with plants is estimated at $60,000 - $100,000 compared to $400,000 for the conventional excavation, and disposal method in USA. In France, the costs for bulk excavation, transportation over short distance and disposal vary from $270 to $460 per ton and long distance transport of excavated soil may be substantially higher (Khalid et *al*., 2017; Chen & Li, 2018). Costs for long distance transport of excavated soil may be substantially higher. Moreover, this technique may not be applicable to agricultural sites because there is a risk of loss of soil fertility.

For developing countries moving towards industrialisation without being aware of the toxicity of metals, the physico-chemical methods are unadapted. Therefore, biological techniques considered as natural, ecological and have no impact on the environment are the most appropriate (Patra et *al*., 2020). Biological remediation approaches involves bioremediation, phytoremediation, bioventing, bioleaching, land forming, bioreactors, composting, bioaugmentation and biostimulation (Kapahi & Sachdeva, 2019; Sayara & Sánchez, 2020; Shah & Daverey, 2020; Da Silva et *al*., 2020). Among these methods, phytoremediation is the most useful (Sarwar et *al*., 2017; Shehata, 2019; Yan et *al*., 2020) and can be a good alternative. It is considered as a novel clean-up option with good public acceptance, less expensive, saving land resources, efficient, environmental and an eco-friendly remediation strategy (Razzaq, 2017; Lajayer et *al*., 2019). Compared to other remediation options, the costs of installation and maintenance are lower for phytoremediation (Wan et *al*., 2016). Islam et *al*. (2014) precised that, the cost for phytoremediation can be 5% less than the cost for alternative clean-up methods. Moreover, phytoremediation allows the restoration of contaminated environments with low costs and low collateral impacts (Ibanez et *al*., 2015).

Since the last two decades, most research efforts have been focused in this field. Phytoremediation approach takes advantage of the unique and the selective uptake capabilities of plant root systems, and applies these natural processes alongside the translocation, bioaccumulation and contaminant degradation abilities of the entire plant (Gomes et *al*., 2016; Yan et *al*., 2020). The diagram presents the phytoremediation technologies involving the removal, containment of contaminants and the physiological processes that take place in plants during phytoremediation (Fig. 7). The advantage of this practice is the minimization of erosion and leaching of soil due to plants coverage (Binzaid & Chowdhury, 2014; Farraji et *al*., 2016).

Phytoremediation includes strategies as phytoextraction (phytoaccumulation), phytostabilization, phytovolatilization, phytofiltration, phytotransformation (phytodegradation) and phytomining (Islam et *al*., 2014; Dhaliwal et *al*., 2020).

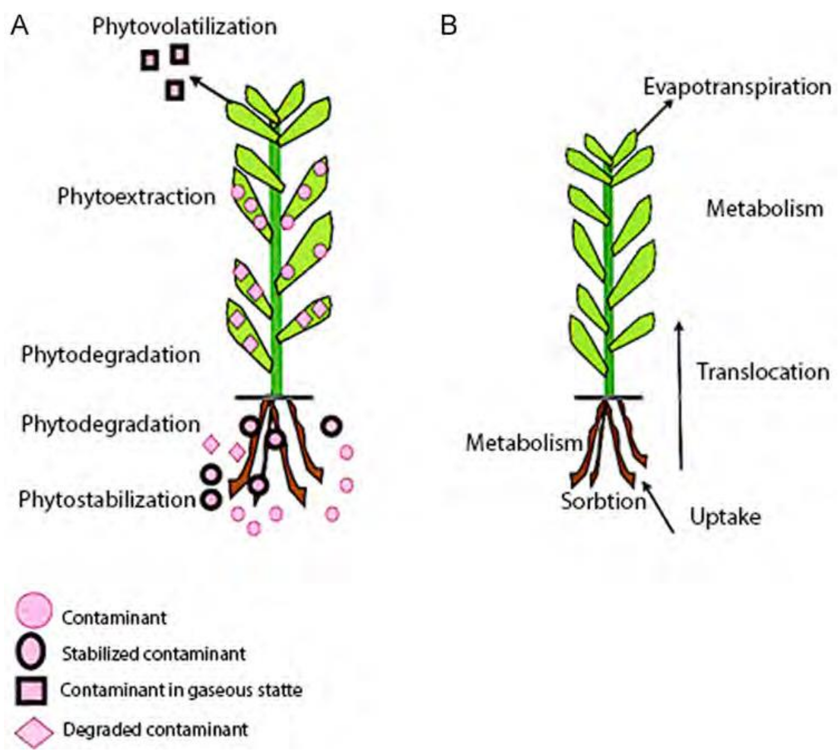

Fig. 7. Diagram of the different phytoremediation strategies (A) and physiological processes taking place in plants during phytoremediation (B) (Gomes et *al*., 2016).

I.2.4.2.2. Phytoextraction

Phytoextraction also known as phytoaccumulation, phytoabsorption or phytosequestration is the uptake of metals from soil or water by plant roots, and their translocation and accumulation in the aerial parts (stem and leaves) i.e., shoots (Ali et *al*., 2013; Ullah et *al*., 2019) (Fig. 8). To preserve soil structure and fertility, it is generally not possible to harvest root biomass. Phytoextration remain a crucial biochemical process because of the translocation of metal from the roots to other parts (shoots) (Mahajan & Kaushal, 2018). This method is the main and most useful technique for the removal of metals and metalloids from contaminated soils, sediments or water, although its efficiency depends on many factors, such as metal bioavailability, soil properties, metal speciation, plant species and, mainly, the concentration of metals in shoots and biomass (Ghori et *al*., 2016; Abdel-Shafy & Mansour, 2018; Lajayer et *al*., 2019). For example, Saad et *al*. (2020) observed on the study of phytoextraction of Pb, Cd and Zn by *Ipomoea aquatica* that this plant has high capacity of translocation of cadmium in particular to shoots.

The phytoremediation potential of plant species is mainly evaluated by two key factors i.e., shoot metal concentration and shoot biomass (Antoniadis et *al*., 2021). According to Suman et *al*. (2018), two different approaches have been tested for phytoextraction of heavy metals:

- the use of hyperaccumulators, which produce comparatively less aboveground biomass but accumulate targeted heavy metals to a greater extent;

- the application of other plant species which accumulates target metals to a lesser extent but produce more aboveground biomass, so that overall accumulation is comparable to that of hyperaccumulators.

However, hyperaccumulation and hypertolerance are more important in phytoremediation than high biomass (Coakley et *al*., 2019; Tatu et *al*., 2020; Balafrej et *al*., 2020). The use of hyperaccumulators is preferable because of the yield of low volume, metal rich biomass which is economical and easy to handle for metal recovery and safe disposal. On the other hand, non-accumulators will produce a high-volume, metal-poor biomass, which will be uneconomic to process for metal recovery and expensive to safely dispose. However, for phytoextraction, grasses are preferable to shrubs or trees because of their high growth rate, better adaptability to environmental stress and high biomass (Pajević et *al*., 2016; Dinu et *al*., 2020). Research has been carried out on the use of crops (such as maize and barley) for phytoextraction with the aim of reducing heavy metals contamination of soils to acceptable levels. The estimation of the phytoextraction duration of a specific heavy metal polluted soil is expressed by a linear relationship between the adsorbed heavy metal contents in the soil and the heavy metal contents in the plant shoots (Cao et *al*., 2018).

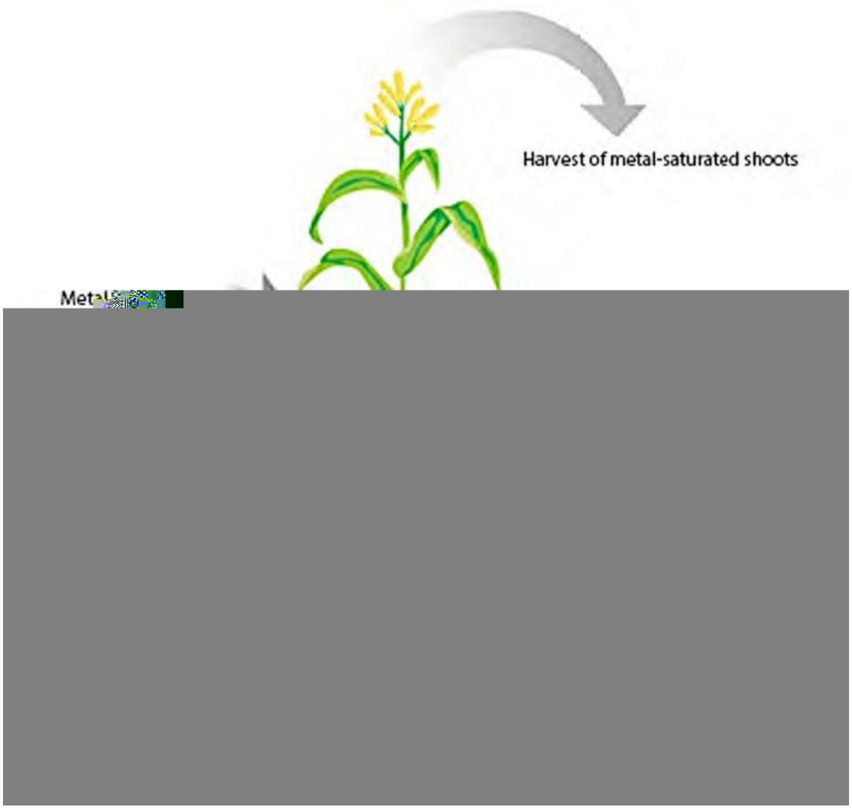

Fig. 8. Phytoextraction diagram (adapted from Nascimento & Xing, 2006)

I.2.4.2.3. Phytostabilization

Phytostabilisation or phytoimmobilization is the use of certain plants for stabilization of contaminants in soils (Radziemska et *al*., 2017). This process can involve simple erosion, leaching or

Euchlaena mexicana, and *Sorghum dochna*) (Zhang et *al*., 2016) and Pb in wetland plants, such as *Juncus effusus* L. (Najeeb et *al*., 2014).

I.2.4.2.4. Phytovolatilization

Phytovolatilization is the mechanism by which plants uptake pollutants from soil, convert them into volatile form and subsequent release into the atmosphere through the stomata, where gas exchange occurs (Limmer & Burken, 2016). Plants can also extract volatile pollutants (e.g., selenium and mercury) from the soil and volatilize them from the foliage (Shahid et *al*., 2017), or in association with microorganisms (plant-assisted bioremediation), as well as degrade organic pollutants. However, the use of phytovolatilization is limited by the fact that, it does not completely remove the contaminant from the environment. The pollutant is simply transferred from one environmental compartment (soil) to the other (atmosphere), from where it can return to the ecosystem via precipitation with rainfall (Gomes et *al*., 2016). This makes phytovolatilization the most controversial of phytoremediation technologies (Limmer & Burken, 2016). A diagram of the phytovolatilization process is displayed in Fig. 10.

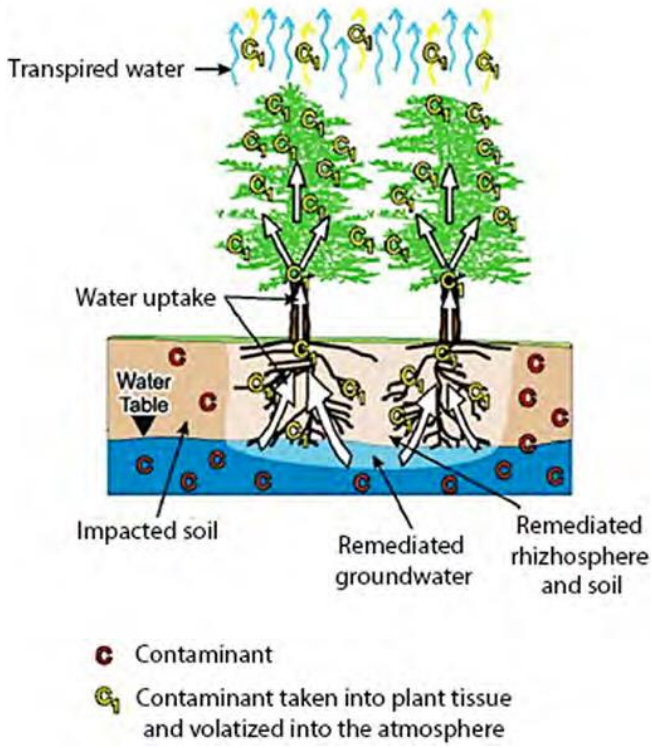

Fig. 10. Diagram of the phytovolatilization process of metals (adapted from Gomes et *al*., 2016).

I.2.4.2.5. Phytofiltration

Phytofiltration is the removal of contaminants through absorption or adsorption from the contaminated surface or wastewater by plants (Benavides et *al*., 2018). This process can be categorized as rhizofiltration (use of plant roots), blastofiltration (use of seedlings) or caulofiltration (use of excised plant shoots; Latin caulis = shoot) (Rezania et *al*., 2016; Ojuederie & Babalola, 2017). It is a system where the movement of pollutants to underground waters is minimized.

Rhizofiltration is the use of plant roots to absorb and adsorb pollutants, mainly metals (especially Pb), through precipitation, absorption and accumulation in the plant biomass. It is mainly applied using aquatic macrophytes, although studies indicate that some terrestrial plants can also practice rhizofiltration, using a root biofilter. Metal precipitation is caused by root exudates which, in turn, change the pH of the rhizosphere (Ali et *al*., 2020). Lead is accumulated in the roots due to certain physiological barriers against metal transport to aerial parts, while other metal such as Cd can easily be moved around in plants. *Pelar gonium* (Arshad et *al*., 2008) and *Brassica napus* (Zaier et *al.*, 2010) are characterized as Pb hyperaccumulators, and they can extract high quantity of lead from contaminated soil without showing morpho-phytotoxicity symptoms (Kumar et *al*., 2014). For most plant species, most of the absorbed lead (about 95% or more) accumulates in the roots, and only a small fraction is transferred to the aerial parts of the plants, as has been reported in *Vicia faba*, *Pisum sativum*, *Phaseolus vulgaris* and *Vigna unguiculata* (Shahid et *al*., 2015; Ukaoma et *al*., 2015). As regards to the aquatic macrophytes, several species such as *Eichhornia crassipes* have shown a high rhizofiltration potential as it is able to absorb high concentrations of Cu, Zn, Ni, Pb, accumulating these elements mainly in roots, with concentrations 3 - 15 times higher than in the shoots (Mishra & Maiti, 2017). In another study, macrophytes such as *Salvinia herzogii*, *Pistia stratiotes*, *Hydromistia stolonifera* and *E. crassipes* were highly efficient in the absorption of Cd, with *P. stratiotes* showing higher growth rates (Yadav et *al*., 2015).

Blastofiltration takes advantage of the sudden increase in surface to volume ratio that happens after germination and the fact that many seedlings are able to adsorb or absorb large amounts of metal, making them uniquely suitable for water remediation (Krishna et *al*., 2012). In the literature, seeds of *Ricinus communis*, *Abelmoschus esculentus*, *Cucumeropsis mannii* and *Moringa oleifera* were investigated with regard to their blastofiltration potential. In water contaminated with 60 ppm of Pb and Cd each, separately, in 72 hours metal content decreased by 96 - 99%. *Ricinus communis* and *Abelmoschus esculentus* seeds were the most efficient, while *Moringa oleifera* seeds removed 100% of Cd from contamined water (Udokop, 2016). In this case, plant seeds could represent the next generation of green technology at bioremediation of heavy metal polluted water with lesser economic importance (Kapahi & Sachdeva, 2019; Da Silva et *al*., 2020). A study of Ravikumar & Sheeja (2013) reported that with the use of aqueous extracts from *Moringa oleifera* seeds, metal uptake from contaminated water as 95% for copper, 93% for lead, 76% for cadmium and 70% for chromium. The results of the use of *Moringa oleifera* cake residues showed the removal of 69.99% Fe, 88.86% Cu, 93.73% Cr, 82.17% and Cd up to 98% and the reduction of 82.17% Pb. These results show that the leaves of *Moringa oleifera* can be used to remove Cd (II) from the synthetic water. Similarly, a study on the removal of heavy metals by a clay-polymer composite of *Moringa oleifera* and bentonite showed a greater potential for metal ions removal (up to 100% reduction) (Ali, 2019; Ravikumar & Udayakumar, 2019). Seeds of *Carica papaya* added to aqueous solutions contaminated with zinc at different pH values showed that Zn absorption increased with contact time and agitation speed of the solutions. While the effective pH of maximum Zn absorption was 5.0, which shows that the efficiency of absorption depends on the pH. In addition, a decrease in sorbent particle size led to an increase in Zn sorption due to the increase in surface area and, consequently, binding sites (Ong et *al*., 2012). *Mangifera indica* seed powder has also been applied for the removal of Cu, Cd and Pb from aqueous solutions, and results indicated the removal ranged from 85% to 100% for all three metals (Parekha et *al*., 2002; Kittiphoom, 2012).

With regards to caulofiltration, recent studies have indicated that *Ipomoea aquatic* showed significant sequestration of excess metal in stem tissue when exposed to Pb concentrations over 20 mg/L. The ability of plant to store Pb in its roots and lower part of the stem coupled with its capacity to produce adventitious roots and lateral branches from the nodes, raises the possibility of using *Ipomoea aquatica* for phytoremediation of Pb in effluents (Chanu & Gupta, 2016). In a study conducted by Hajar et *al*. (2014), excised stems of *Stevia rebaudiana* accumulate significant amounts of As, Cu, Se and Al, while excised shoots accumulate significant amounts of Cd, Ni, Pb or Zn in the leaves. However, the application of the process of metal phytoremediation is not as widespread (Gomes et *al*., 2016).

I.2.4.2.6. Phytotransformation

Phytotransformation or phytodegradation, refers to the uptake of pollutants and nutrients from water, sediment or soil and their chemical modification as a direct result of plant metabolism. This process often results in the inactivation, degradation or immobilization of contaminants (Tangahu et *al*., 2011; Gomes et *al*., 2016), and occurs both in the roots (rhizodegradation) and/or shoots (Bulak et *al*., 2014). It involves the use of plants for the degradation of organic contaminants using enzymes such as dehalogenase and oxygenase (plant-assisted bioremediation) (Kumar & Singh, 2018). Enzymes produced by plants are used to metabolize toxic elements and convert them into less toxic compounds. Microorganisms such as bacteria, yeasts and fungi also contribute to this process (Ojuederie & Babalola, 2017; Li et *al*., 2020). Therefore, this process usually occuring for organic compounds is not dependent on rhizospheric microorganisms (Schwitzguébel, 2017). Recently, scientists have directed their interest in studying the phytodegradation of organic pollutants such as synthetic herbicides and insecticides as heavy metals are non biodegradable. For this purpose, genetically modified plants can be useful (Saxena et *al*., 2019).

Rhizodegradation refers to the degradation of organic contaminants in the soil by micro-organisms in the rhizosphere (Correa-García et *al*., 2018; Allamin et *al*., 2020). The rhizosphere is under the influence of the plants and extends about 1 mm around the roots (York et *al*., 2016). Plants can stimulate an increase in the numbers and metabolic activities of microbes, about 10–100 times higher in the rhizosphere by the secretion of exudates containing carbohydrates, amino acids and flavonoids. Nutrient-containing exudates released from plant roots provide carbon and nitrogen sources to the soil microbes and create a nutrient-rich environment where microbial activity is stimulated. In addition, plants can also release certain enzymes capable of degrading organic contaminants in soils to facilitate the growth and activities of rhizospheric microorganisms (Gkorezis et *al*., 2016; Ojuederie & Babalola, 2017).

I.2.4.2.7. Phytomining

Phytomining is an emerging approach to detect the reserve of valuable elements (like gold) in a particular underground place (Harumain, 2016). It is a useful technique particularly for removing metals from soil. There are some plants hyperaccumulators, that can accumulate rare metals such as gold and nickel from soil in their harvestable parts, and these metals can be extracted from plants and recovered through incineration (Suman et *al*., 2018).

I.2.4.3. Phytoremediation approach

A key step in the phytoremediation of heavy metal contaminated soils is the screening of hyperaccumulators and accumulators (Sarwar et *al*., 2017; Méndez et *al*., 2018). Phytoremediation included two approaches:

- the use of hyperaccumulators that can relatively produce a low amount of above-ground biomass but accumulate a high amount of one or more elements;
- the application of high biomass that produces plants that are characterized by a lower capacity to accumulate targeted elements. Here, due to the high yield of above-ground biomass, total element uptake is comparable to that of hyperaccumulators (Chandra et *al*., 2018).

Hyperaccumulators are macrophytes capable to accumulate over 1000 $mg.g^{-1}$ heavy metals in in their shoot tissues (without visible toxicity symptoms) (Suman et *al*., 2018; Coakley et *al*., 2019). However, different researchers have defined hyperaccumulators as plant species capable to accumulate 100 – 500 fold higher metals in shoots with no effect on the yield as compared to common nonaccumulator plants (Mahar et *al*., 2016; Sheoran et *al*., 2016). Storage and accumulation requirements of hyperaccumulator plant species are different for different metals. Plant species which accumulate > 100 $mg.kg^{-1}$ Cd and Se (on dry weight basis), > 1000 $mg.kg^{-1}$ Cu, Ni, As, and Pb or > 10,000 $mg.kg^{-1}$ Mn and Zn in their aerial plant parts when grown on heavy metal(loid) contaminated soils are called hyperaccumulator plants (Mahar et *al*., 2016). Currently, more than 450 hyperaccumulating plant species of 45 families fulfilling the criteria of being hyperaccumulators are known, which represents less than 0.2% of all angiosperms, the majority of them being Ni hyperaccumulators (75%) (Messou, 2015; Adesuyi et *al*., 2018; Eid et *al*., 2020). For hyperaccumulators, a prerequisite is to tolerate efficiently high concentration of metals within plant tissues and cells, although the standard for hyperaccumulators has not been defined scientifically (Khalid et *al*., 2017).

For phytoaccumulation (phytoextraction), grasses have a higher preferable in use than shrubs or trees because of their higher growth rate, more adaptability to stress environment and high biomass (Ghori et *al*., 2016). The use of hyperaccumulators to decontaminate polluted soils might result in production of a bio-ore of some commercial value to recoup some of the costs of soil remediation (Méndez et *al*., 2018).

I.2.4.4. Variables influencing metal phytoremediation processes

Several factors can influence the heavy metal uptake mechanism and its efficiency, including plant species, environmental properties, microorganism-plant interactions and translocation, tolerance mechanisms, soil and metal characteristics (Mahar et *al*., 2016) (Fig. 11). Concerning plant species, because of their contaminant uptake characteristics, they are selected as species with a high remediation potential (Adesuyi et *al*., 2018). However, for phytoextraction, their efficiency is directly influenced by the bioavailability of metals. Low bioavailability is a major limiting factor for phytoextraction of contaminants such as Pb. Usually, only a small fraction of metal in soil is bioavailable for uptake by plants (Petruzzelli et *al*., 2015). Metals bind strongly to soil particles or precipitation makes a significant fraction of the heavy metals in the insoluble (Mahar et *al*., 2016; Sheoran et *al*., 2016). However, plants have developed certain mechanisms for solubilizing heavy metals in the soil, such as the secretion of metal-mobilizing substances in the rhizosphere, called phytosiderophores (Benizri & Kidd, 2019). In addition, secretion of H^+ ions by roots can acidify the rhizosphere and increase the dissolution of metal, since H^+ ions can displace metal cations adsorbed on soil particles (Ma et *al*., 2016). The pH of the soil rhizosphere can be lowered by root exudates by one or two units generally higher than in bulk soil. Therefore, lower soil pH can increase the concentration of heavy metals in solution by promoting their desorption (Qiang et *al*., 2018). Furthermore, the rhizospheric microorganisms (mainly bacteria and mycorrhizal fungi) can significantly increase the bioavailability of heavy metals in soil (Ma et *al*., 2016; Correa-García et *al*., 2018). The chemical composition and sorption properties of soil influence the mobility and bioavailability of metals (Ahmed et *al*., 2020). With regards to the bioavailability of heavy metals/metalloids in soil, there can be three categories: readily bioavailable (Cd, Ni, Zn, As, Se, Cu); moderately bioavailable (Co, Mn, Fe) and least bioavailable (Pb, Cr, U) (Tangahu et *al*., 2011).

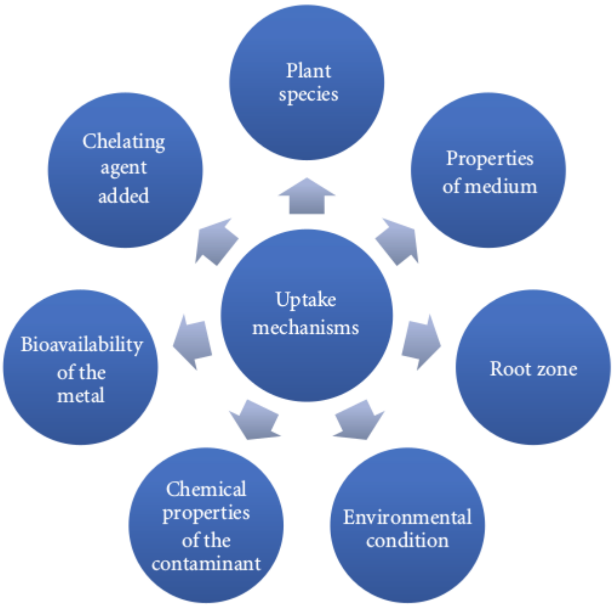

Fig. 11. Factors affecting the uptake mechanisms of heavy metals (Tangahu et *al*., 2011).

Bioavailability can be increased by lowering soil pH, as metal salts are soluble in acidic rather than in basic media (Petruzzelli et *al*., 2015; Kee et *al*., 2018). Factors such as pH, oxygen content, water availability, temperature and organic matter influence the valence of metals in water or soil. In the study carried out by (Bauddh & Singh, 2012), it was found that *Ricinus communis* tolerated salinity and drought better in the presence of Cd and removed more Cd in a given time than Indian mustard. These species also produced a significantly higher biomass than *Brassica juncea* when grown in soil contaminated with Cd in the presence of 100 mM NaCl salinity and after a ten-day water withdrawal, indicating the importance of these variables in phytoremediation processes.

In sediments, electrical conductivity and pH can cause changes in metal speciation and solubility, which can lead to a flow of metals from the pore water to the water column and/or increased uptake by plants (Caporale & Violante, 2016; Kee et *al*., 2018). It has been reported by Khellaf & Zerdaoui (2013) and Caporale & Violante (2016) that these factors directly attempt at phytoremediation with *Lemna gibba*, which observed that this macrophyte has a great potential to remove Zn from contaminated waters, especially at 21 °C and a pH between 5 and 6. However, development of these species at temperatures of 17, 25 and 29 °C and at pH values between 3 and 4 favors negative effects on the plant and does not favor the absorption of Zn. In another study, carried out by Kee et *al*. (2018), higher temperatures and lower soil pH led to a significant increase in cadmium and zinc. However, studies on phytoremediation and factors influencing metal uptake in plant species done by Sheoran et *al*. (2016), Kee et *al*. (2018) reported that the addition of lime and lignite (sedimentary rock) to polluted soil reduced the uptake of cadmium and zinc by plants due to increase in soil pH, with no difference in the uptake of copper or lead. It has also been showed that the presence of organic matter modifies the efficiency of the phytoextraction of certain elements. For example, in a study conducted with *Ricinus communis* L. (castor), the effect of adding organic matter (peat) to soils contaminated with metals and boron (B) was evaluated, and it was observed that plants grown without organic matter showed no accumulation of Cr, Ni, Cd, Cu, Pb and Zn, while B concentrations increased (Abreu et *al*., 2012).

I.2.4.5. Fate of plants after phytoremediation

After the study on the phytoremediation, an interesting question remains: what will be the fate of plants after being used for phytoremediation of heavy metals? This because, the metal accumulation and their removal by aquatic plants would not be enough if there is a lack of proper management for a successful implementation. This can be outlined in Fig. 12. There may be some processes for the disposal of these aquatic plants, but it is difficult to elucidate whether this would be feasible or not in economical and environmental plan. However, some research show the possible ways to handle the harvest plant used for phytoremediation of heavy metals (Ali et *al*., 2013, Nzihou & Stanmore, 2019).

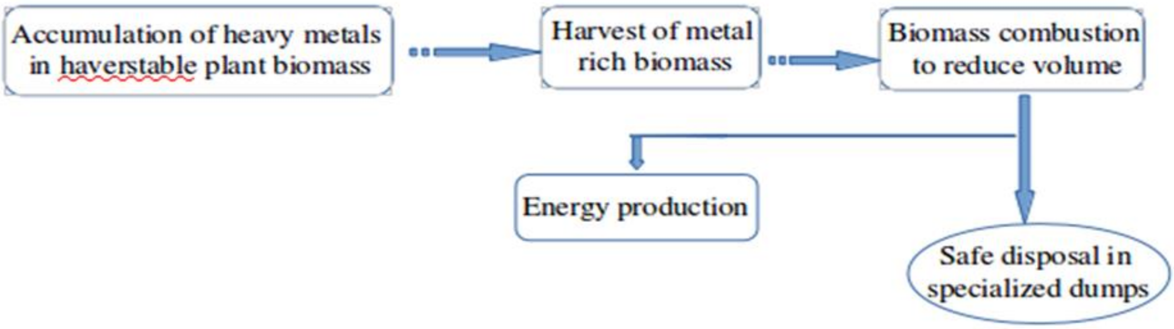

Fig. 12. Post-harvest treatment of phytoremediator plants (adapted from Ali et *al*., 2013).

I.2.4.5.1. Carbonization and incineration

Aquatic plants with a high metal content can be used to make charcoal by incineration and the derived gas can be used as fuel. For this purpose, *Eichhornia crassipes* (water hyacinth), *Pennisetum purpureum* (elephant grass) has been used (Danquah et *al*., 2018; Wang et *al*., 2019). However, fresh aquatic plants take longer time for drying and have high moisture content and there is no evidence whether after burning the plants, the pollutant is completely vanished. Carbonization of the plants with high heavy metal contents may also be a source of toxic emission in the air. It has been reported that burning high As-containing coal is one of the major sources of As exposure (10 - 20% of total As exposure) for the population of Guizhou, China (Wang et *al*., 2019). Another study also revealed that burning coal with high arsenic content increased arsenic content in hair, urine, and blood in children residing in polluted areas (Feng et *al*., 2015; Sun et *al*., 2017; Curtis et *al*., 2018). Thus, the burning of hyperaccumulative aquatic plants would not be harmless to the environment, and would be dangerous to human health.

I.2.4.5.2. Hydrolysis and fermentation

Liquid fuel, such as ethanol, can be produced in aquatic plants during phytoremediation by hydrolysis at the same time as fermentation. Hydrolysis and fermentation also require fermentable yeast sugars that may be available only to a small extent in the aquatic plants for phytoremediation. Certain types of pre-treatment are, therefore, required to make the sugar more readily available for chemical hydrolysis (Loow et *al*., 2016; Kucharska et *al*., 2018). Pre-treatment requires relatively high temperature, strong acids and pressurized reactors. Rezania et *al*. (2015), Bušić et *al*. (2018), Singh et *al*. (2017) concluded that hydrolysis of water hyacinths to produce fuel is only possible in situations where there is a high need for ethanol as a liquid fuel due to the negative energy balance. Although, it is economically feasible to produce fuel from aquatic phytoremediating plants, the heavy metal content of the by-product sludge and its potential for recontamination should be tested.

I.2.4.5.3. Briquetting

Briquetting would be a good option for the treatment of the aquatic phytoremediating plants. Briquettes have been widely sold commercially for cooking food. Carnaje et *al*. (2018), Okwu et *al*. (2016) reported that briquetting was a possible treatment of water hyacinth. Briquettes are made by drying the water hyacinth in the sun for a few days, disintegrating, screening and cutting the dried water hyacinth into pieces of about 6 mm length. The shredded water hyacinth can then be compressed into briquettes or pellets. The material obtained after briquetting the water hyacinth has an energy density of 8.3 GJ m^{-3}, which is comparable to charcoal, which has a density of 9.6 GJ m^{-3} (Gunnarsson & Petersen, 2007; Carlini et *al*., 2018).

I.2.4.5.4. Bio-recovery or disposed as hazardous waste

Phytoremediation plants after combustion can either be safely disposed of as hazardous waste in specialized landfills like other hazardous materials or if economically feasible, treated for bio-recovery of precious and semi-precious metals. This practice is known as phytomining (Wang et *al*., 2019) (Fig. 13). Plant biomass containing accumulated heavy metals can be burned for energy. The remaining ash is considered "bio-ore". This bio-ore can be processed for the recovery or extraction of the heavy metals. The commercial viability of phytomining depends on many factors such as the efficiency of phytoextraction and the current market value of the metals treated.

Phytomining has been used commercially for Ni and is believed to be less expensive than conventional extraction methods. Using *Alyssum murale* and *Alyssum corsicum*, a biomass containing 400 kg Ni ha^{-1} can be grown with production costs of \$250 - 500 ha^{-1}.

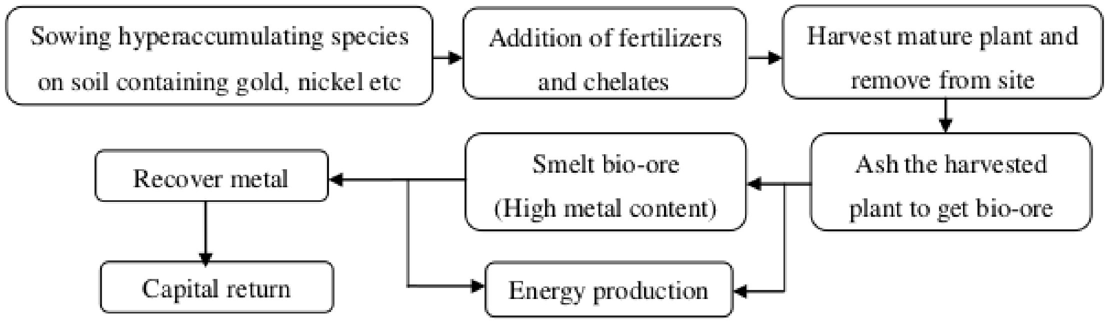

Fig. 13. Integrated process of metal recovery or phytomining (modified from Sheoran et *al*., 2009).

I.2.4.6. Advantages, limitations and future perspectives of phytoremediation

Phytoremediation is effective for treatment of large areas, efficient, eco-friendly and based on solar technology. It is also considered as an important tool in ecological engineering and its main advantages are low installation and maintenance costs compared to other remediation options (Patra et *al*., 2020). It is popular with the public, who accept it as a "green clean" alternative to chemical plants (Shehata, 2019; Shah & Daverey, 2020). In addition, the use of plants in synergistic phytoremediation techniques does not only clean the environment, but also restores ecosystems (Shehata, 2019; Shah & Daverey, 2020). From an economic point of view, the phytoextraction of metals with a market value such as nickel (Ni), thallium (Tl) and gold (Au), can be removed and used, sold or recycled for sustainable land management. This technique can gradually improve soil quality for the subsequent cultivation of higher value crops (Méndez et *al*., 2018; Sricoth et *al*., 2018). In addition, depending on the quality of plant biomass after phytoremediation and chemical treatment for decontamination, they can also be used for energy production (biogas or direct combustion), ethanol and brick production, and paper manufacturing (Carlini et *al*., 2018; Bušić et *al*., 2018). For example, the use of *Eichhornia crassipes*, an aquatic macrophyte, has been applied in the manufacture of building bricks (Carnaje et *al*., 2018).

Although, phytoremediation has many advantages, it also suffers from certain limitations. The use of plants to clean up the environment often takes longer than other remediation techniques and is best suited for areas where the elements are present in the plant root zone (Ashraf et *al*., 2019; Ali et *al*., 2020). In addition, as previously mentioned, environmental conditions are a determining factor in the efficiency of phytoremediation, and may not always be adequate for most species. However, soil contamination by multiple metals requires the use of specific species that are well-adapted or tolerant to the environmental conditions and the contamination present, and allow for a positive synergistic interaction between plant roots to achieve and tolerate the negative effects caused by metals zone (Yan et *al*., 2020; Shah & Daverey, 2020). Thus, the application of phytoremediation in these cases also requires a wide range of research prior to the application of the technology (Khalid et *al*., 2017). The bioavailability of the metal is also an issue; for example, if the metal is closely bound to the organic parts of the soil, it may not be bioavailable, whereas, if the metal is water-soluble, it will pass through the root without accumulating in the plant (Petruzzelli et *al*., 2015). Phytoremediation approach is a relatively promising area of research, which is currently limited to laboratory and green house scale studies due to the above-mentioned limitations, and only a few studies have been conducted to test the potential of phytoremediation on the field. Many factors can influence phytoremediation on the field, including variations in temperature, nutrients, precipitation and moisture, plant pathogens, uneven distribution of contaminants, soil type, soil structure, soil/water pH, and redox potential (Mahar et *al*., 2016; Sheoran et *al*., 2016) and other environmental conditions. However, the effectiveness of phytoremediation of different plants for specific targeted heavy metals needs to be tested under field conditions in order to evaluate the feasibility of this technology for practical commercialization.

CHAPTER II. MATERIAL AND METHODS

II.1. Material

II.1.1. Description of the study area

This study was carried out from September 2016 to January 2020 during the long rainy season (RS) and long dry season (DS) in the city of Yaounde (Cameroon) and its surroundings. Yaounde, in the center Region was an urban area of approximately 256 km^2, was located between latitudes 4°45' N and 4°00' N and longitudes 11°20' E and 11°40' E with an average altitude of 750 m above the sea level. Yaounde, had a sub-equatorial guinean warm and humid climate with average annual temperature of 23.5°C. The local bimodal climate was composed of four seasons, two rainy and two dry organized as follows: a long rainy season from mid-August to mid-November, a long dry season from mid-November to February, a short rainy season from March to June and a short dry season from July to August (Zogning et *al*., 2011; Abossolo et *al*., 2015). The mean annual rainfall was about 1600 mm per year. Due to its hilly relief, and its important drainage network, Yaounde was called the "city of seven hills" and had many lowland areas. These included depressions drained by streams or rivers, more or less abandoned lakes and fishponds created by drainage of stream water or fed by groundwater and depressions temporarily flooded during peak rainy season and after individual thunderstorm events. All were located in areas where rapid densification of urban landuses is increasing pressures on these environments (Onana et *al*., 2005; Nguyen et *al*., 2019). The hydrographic network was moderate, with many small catchment areas that drained into the Mfoundi watershed. The primary vegetation, which was formerly equatorial forest, has been transformed by urbanization into tertiary forests. The geology of Yaounde revealed the existence of schists, quartzites, gneisses and migmatitic gneisses reaching the granulite facies (Tchakounte et *al*., 2017; Ngamy et *al*., 2019). Yaounde had ferrallitic soils which were very thick and composed of kaolinite, gibbsite, goethite, hematite, quartz, accessory rutile, zircon, and magnetite, rich in iron and aluminum oxides, very fertile but extremely fragile but rocks into ferrallitic and pseudogley soils were weathered by the climate (Djoufack, 2011; Ndjigui et *al*., 2013). The population of Yaounde city was estimated at 2.4 million inhabitants in 2011 (Anonymous 9, 2012). Yaounde faces overpopulation like many other urban cities in developing countries with a density of 14,000 inhabitants/km^2. This situation contributed the increased frequency of flooding in urban areas (Zogning et *al*., 2011).

II.1.2. Localization of the study sites

Sites were selected according to criteria such as anthropogenic and industrial activities of which discharges were polluting. Polluted sources sites include, but are not limited to extension of urban agriculture, proliferation of sewage and solid wastes from households, commercial activities (repairs shops, garages, paintings, hostels, and markets), wastes from administrations, proliferation of solid dumpsites, pits and septic tanks, sewage from treatment plants and hospital effluents. Area free of any potential activity served as a control site. Based on the above criteria, twelve (12) periodically inundated lowlands sites were selected and Table IV presents the description of each site (Appendix 8). Among them, 11 lowlands areas sites were chosen in the urban area of Yaounde and 01 unpolluted site (control) in a village located at about 25 km of Yaounde (Fig. 14).

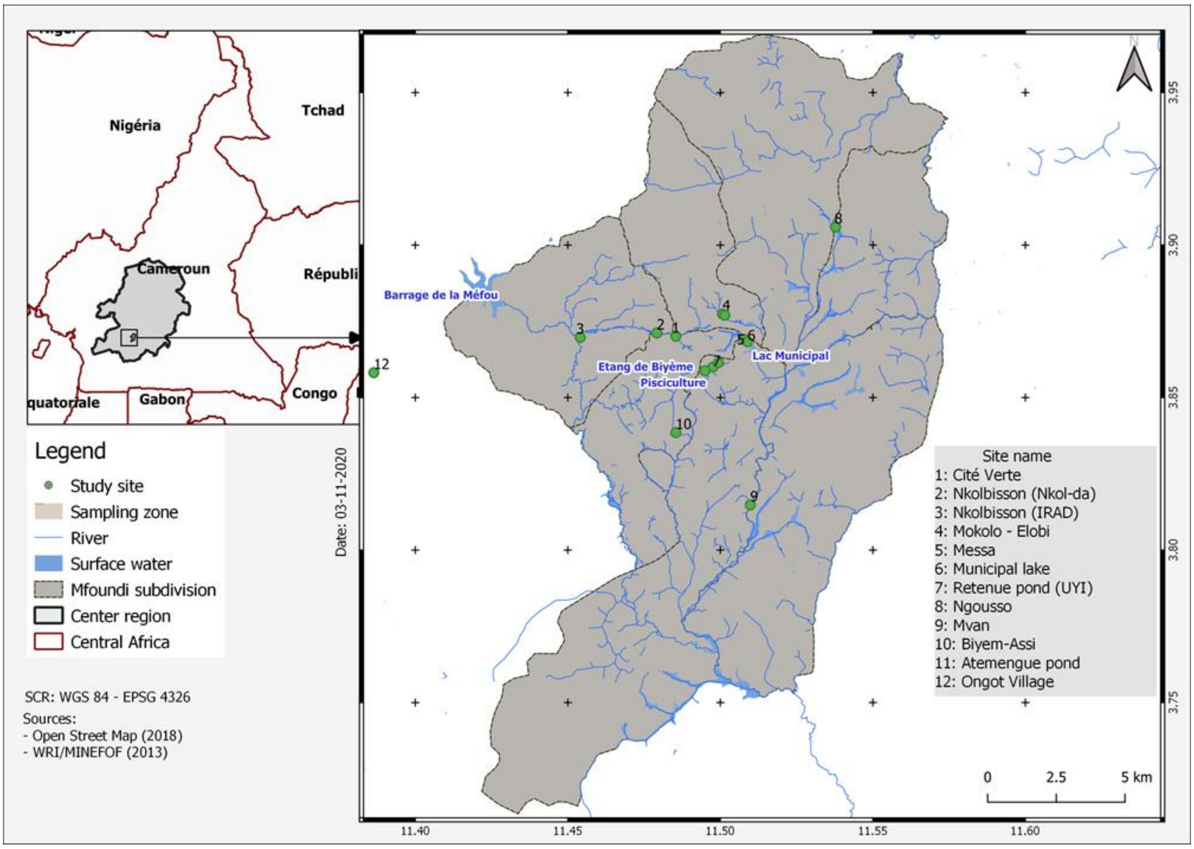

Fig. 14. Location map with drainage network

II.2. Methods

II.2.1. Identification of the pollutant-tolerant flora colonizing lowlands of Yaounde

II.2.1.1. Floristic survey method

For macrophyte species cartography, prospection studies were carried out on each site. In the 12 lowland sites, each pollution source was geo-referenced with a hand-held GPS receiver (Garmin GPSMAP 64st). Sites were grouped around the city center of Yaounde in all directions so that seven (7) from 12 belong to the Mfoundi watershed to have a representativeness of lowland flora and 5 others belong to the Ntem, Olezoa, Biyeme and Ogok Watersheds.

At each lowland site, the macrophyte species inventory survey varied according to the length of the watercourse, the identified sources of pollution, the position and the form of the floodplain. According to this classification, three types of lowland surfaces were identified: flat, undulating and longitudinal. The surveys were made in all twelve lowland sites chosen according to the quadrat sampling method with 1 × 1 m square (Gillet, 2000).

Table IV. Description of lowland sites and the surrounding activities sources of contamination

Sites	GPS coordinates & names	Description of lowlands and surrounding activities sources of the potential contamination	Lowland types /watershed
Site 1 Cité-verte	X: 03°52'12.3" Y : 011°29'07.5" Z : 708 m	This site was located below the wastewater treatment plant of the Cite-verte where treated effluents and wastewater drained through sewers and gutters were discharged. This watercourse measures approximately 300m long and the water flows until it discharges into the Abiergue watershed. It was surrounded by activities such as faecal sludge treatment plant, commercial activities, fish farming, household sewage.	Lowland/ Abiergue
Site 2 Nkol-nso'o (New road Nkolbisson	X: 03°52'16.1" Y: 011°28'45" Z: 721 m	This site was the drop-off point between the Oyom-abang neighborhood and the new Nkolbisson-road. The predominant pollution was that of hydrocarbons which were dumped and drained into the watercourse. There were garages, buildings, a fleet of old vehicles and used oils.	Lowland/ Abiergue
Site 3 Nkolbisson-IRAD	X: 03°52'11" Y: 011°27'14.6" Z: 708 m	This site was a natural depression in the lowlands of the Abiergue watershed. It was approximately 2 ha in size and was located below the IRAD of Nkolbisson. The predominant pollution in this environment was organic, chemical and metallic trace elements (MTE) and come mainly from houses and some commercial structures along the basin (garages, laundries, cinder block factories, agriculture with the use of chemical fertilizer, insecticides, and pesticides, laboratories, poultries and farms).	Lowland/ Abiergue
Site 4 Mokolo-elobi	X: 03°52'38.5" Y: 011°30'02.6" Z: 729 m	This site measured approximately 3 ha in area, runoff, and water from agricultural activities were drained from it. This site was very flooded during the rainy season, while only the furrows retain water during the dry season. The site was totally covered by macrophytes and market gardens on 1/4 of its surface. Pollution was increasing due to organic amendments (chicken droppings). The market, agricultural activities, leachate solid-dumpsites, garages, paintings, repairs-shops, sceptic pits were the main activities around the site.	Lowland/ Abiergue
Site 5 Messa	X: 03°52'08.6" Y: 011°30'28.7" Z: 725 m	This site was located below the subsurface water treatment plant of the Messa camp-SIC, which drained treated and untreated effluent and storm water into the municipal lake where they were discharged into the Mingoa watershed. The latter was about 1ha in size and was congested by fields, cinder block factories, wastewater treatment pond, household sewages, agricultural activities, hospitals, buildings, and garages.	Lowland/ Mingoa
Site 6 Municipal lake	X: 03°52'05.8" Y: 011°30'32.2" Z: 727 m	This site was located in the heart of the city of Yaounde and was the result of a dam built in the following areas on the Mingoa River, a tributary of the Mfoundi river. Covering an area of approximately 25.000 m^2, this area received effluent from the mini-step of the Yaounde deputies' hostel, the Messa step, upstream housing, Deputy hostel, restaurants, household waste from the buildings, effluent from the wastewater treatment pond, and garages.	Lake/ Mingoa

Site 7 Retenue pond (UYI)	X: 03°51’35.8” Y: 011°29’51.2” Z: 724 m	This site was located below the university buildings of Ngoa-ekele, National School of Polytechnic, CHU and where the agricultural activities were developed and drains the wastewater from these buildings to the Atemengue lake located downstream of the latter. This lake measured approximately 1.5 ha and was currently eutrophied. The residents use 2/3 of the site for agriculture and the students for their academic experiments. The pollution was rather organic, chemical and metallic due to the use of chemical fertilizer in agriculture, chemical reagents used in laboratories.	Pond/ Olezoa
Site 8 Ngousso	X: 03°54’21.2” Y: 011°32’16” Z: 728 m	This site was located at the northeastern end of Yaounde, below the Yaounde General Hospital and the Yde Obstrical-Genic Hospital and near the railroad. Pollution by hydrocarbons, MTE from used oils, as well as organic and chemical pollution were observed. This shallow area drained water that flows into the Mfoundi. With a surface area of about 50 ha, houses, rubbish bins, hospitals wastewater, garages, car washes, livestock farms, pit latrines, septic-tanks, railway oil and solid leachate dumps.	Lowland/ Ntem
Site 9 Mvan	X: 03°48’53.1” Y: 011°30’35.3” Z: 688 m	This site was located at the southeastern edge of the city of Yaounde, where most of the effluent from the brewery industries was drained, and where housing and cinderblock factories were developed. Organic, chemical and metallic pollution seems more present at this site. Agroindustries (SOFAVIN), buildings and garages were activities that were developed there.	Lowland/ Mfoundi
Site 10 Biyem-assi	X: 03°50’18.6” Y: 011°29’07.5” Z: 705 m	This site was located at the southwestern edge of the city of Yaounde, below the Biyem-assi water treatment plant that drained treated and untreated water. With a surface area of about 1.8 ha, organic, chemical and MTE pollution was likely to occur due to surrounding activities such as faecal sludge treatment plant, garages, paints, domestic sewage effluents, leachate solid-dumps, and livestock.	Lowland/ Biyeme
Site 11 Atemegue pond (Obili)	X: 03°51’41” Y: 011°29’57.7” Z: 736 m	This pond, dredged from the Olezoa stream for fish farming purposes, was located at the bottom of the Yaounde I University and received untreated effluent from the university campus, WWTP, agriculture, CHU hospital and the mortuary, laboratories of the UYI, FMBS of Yaounde and fish farming. It measured about 2.7 ha and this area was in a very high state of eutrophication and was covered with macrophytes. It drained the water from the mortuary to the Olezoa basin. The fishponds (3 ponds) were connected to the site and there were large-scale fish farming structures at about 10 meters with 6 ponds per tank of 2 to 3 m^3 each.	Pond/ Olezoa
Site 12 (Control) Ongot village	X: 03°51’29.3” Y: 011°23’10.9” Z: 735 m	Site 12 (Control) was located on the northwestern outskirts of Yaounde. It was a vast floodplain, which was largely drained by the waters of the Ogok river. The vegetation was pioneer because no anthropic activities have developed in this area.	Lowland/ Ogok

At each site, transects of 50 m was randomly established on either side of the watercourse, comprising five quadrats of 1 × 1 m square was set-up for the longitudinal lowlands (Appendix 1). For flat and sinusoidal inland valleys, a variable number of quadrats were randomly installed. A total of 231 and 187 quadrats were sampled respectively in the rainy and dry seasons. In each of the quadrats, data such as specie diversity present, their recovery rate, abundance and dominance were determined. A complete list of macrophyte species was drawn up for each of quadrat (Appendix 5). The scientific names of these species were either determined directly in the field using the Adventrop (Weeds of Sudano-Sahelian Africa) as described by Le Bourgeois & Merlier (1995), or by reference to the National Herbarium of Cameroon. Each macrophyte specie recorded was assigned a coefficient expressing its abundance-dominance using the mixed Braun-Blanquet (1964) scale and the Massens index (1997), widely used in the field of plant ecology (Gillet, 2000; Dufrêne, 2003; Meddour, 2011). This scale consists of classed data (from 1 to 5) and characters (C and r) defined by recovery percentages as follow:

- 5: – Individuals – covering> 3/4 of the reference surface (> 75%)
- 4: – Recovery between 1/2 and 3/4 (50–75% of the reference surface)
- 3: – Recovery between 1/4 and 1/2 (25–50% of the reference surface)
- 2: – Recovery between 1/20 and 1/4 (5–25% of the reference surface)
- 1: – Recovery <1/20, or scattered individuals covered until 1/20 (5%)
- C: – Few individuals with very low recovery
- r: – rare.

II.2.1.2. Determination of the macrophytes diversity indices

Any good phytoremediation strategy begins with an ecological analysis of plants (Messou et *al*., 2013). Diversity data collected were used to characterize the flora in the Yaounde lowlands. The diversity of macrophytes in the study sites was evaluated using the Species richness (S), Shannon diversity (H'), Evenness equitability (E), Simpson (D) and Pielou equitability (J') and Sorensen index (K) indices.

Shannon diversity index (H') expresses the diversity by taking into consideration the number of species and the abundance of individuals within each of these species (Frontier et *al*., 2008; N'da et *al*., 2008). It is the recommended index in comparative stand studies, as it is independent of the size of the study population (Yao et *al*., 2010). Thus, a community controlled by a single species will have a lower coefficient than a community of which all species are codominant. The value of the Shannon index varies from zero (a single species) to log S (when all species have the same abundance). Shannon diversity index is low when a single specie is encountered. It is expressed in bit/individual (Frontier et *al*., 2008; N'da et *al*., 2008) and calculated as follows:

$$\boxed{H' = -\sum_{i=1}^{s} Pi \; logPi} \qquad (1)$$

where S is the total number of taxa and Pi ($Pi = \frac{ni}{N}$), is the proportion of individuals in taxon I (ni) in relation with the total number (N) of individuals.

The formula for calculating the Evenness index (E) is:

$$\boxed{E = \frac{H\prime}{\ln(N)}} \qquad (2)$$

where H’ is the Shannon index value and N is the total value of all the species.

Simpson's diversity index is a measure of diversity, which takes into, accounts both richness and evenness. The equation used to calculate Simpson’s index is:

$$\boxed{D = \sum(Pi)^2} \qquad (3)$$

where, D = Simpson index of dominance, Pi = the proportion of important value of the ith species (Pi = ni / N, ni is the important value index of ith species and N is the important value index of all the species).

As D increases, diversity decreases and Simpson’s index of diversity (D’) is therefore usually expressed as: D’= 1 – D or 1/ D.

Pielou’s equitability index (J') is used to measure the distribution of species within site, regardless of species richness (Frontier et *al*., 2008; N’da et *al*., 2008). Low equitability represents the great importance of some dominant species (Ngueguim et *al*., 2010). Pielou’s equitability is low when few species are encountered. Its value varies from zero (dominance of one of the species) to 1 (equitable distribution of species). It corresponds to the ratio between the observed diversity and the S number of species present in the plot:

$$\boxed{J' = \frac{H\prime}{H\prime max}} \qquad (4) \qquad \text{where } H'max = logS.$$

Sorensen index (K) was used to understand the flora community's similarity to the sites. It was used to compare the control and potential contaminated sites in all investigated sites. When K> 50%, it means that the control and the polluted sites considered are floristically similar. On the contrary, when K<50%, it means that the control and polluted sites are floristically differents. K is calculated with the formula:

$$\boxed{K = \left(\frac{2C}{A+B}\right) \times 100} \qquad (5)$$

where A is the total number of species in the control site; B the total number of species in polluted sites; C the total number of species common to both sites.

II.2.1.3. Identification of potential phytoremediating plants of heavy metals polluted lowlands

The identification of potential phytoremediation plants in the lowlands polluted by heavy metals was based on the following criteria: high growth rate, production of above ground biomass, widely distributed and highly branched root system, good adaptation to prevailing environmental and climatic conditions, hyperaccumulation and hypertolerance, resistance to pathogen and pest, easy for cultivation and harvest, high translocation factors (TF) and high bio-concentration factor (BCF) (Khalid et *al*., 2017; Shah & Daverey, 2020). These authors specify that for the phytoremediation process, the selection of potentially usable species should be based on species that demonstrate characteristics of adaptation to the environment, stress conditions and specificities in the phytoremediation strategy. For this purpose, the relative frequency of species and their local coverage were used.

Relative frequency (Fr i) provides information on the rate of occurrence of a species in a site (Gillet, 2000; Ray & Georges, 2009) using the formula:

$$\boxed{Fri = \frac{Fa}{nr} \times 100} \qquad (6)$$

where Fr i is the relative frequency of the taxon i; Fa, the absolute frequency of taxon i. The absolute frequency (Fa) is the number of quadrats in which the taxon i is present. The total number of quadrats is noted nr.

Relative abundance (A) gives the abundance of a species in a site (Meddour, 2011). To calculate the relative abundance of a given species in a table of association, each abundance-dominance (AD) is transformed into percent of average recovery (R), using the scale introduced by (Dufrêne, 2003) (Table V), divided by the number of quadrats (nr) and multiplied by 100 according to the equation below:

$$\boxed{A = \frac{(\sum R \times 100)}{nr}} \qquad (7)$$

Table V. Correspondence between abundance-dominance (AD) coefficients and average recovery (R) values (Dufrêne, 2003).

AD	5	4	3	2	1	+	r
R	75-100 87,5	50-75 62,5	25-50 37,5	5-25 15	01-5 2,5	<1 0,2	/ 0,1

II.2.2. Assessment of the level of heavy metals contamination in soils, water and plants

II.2.2.1. Choice of study sites and plants species

Among the 12 lowland sites, three study sites were selected according to the abundance of pollutant-tolerant plant species (site 4: Mokolo-elobi, site 9: Mvan and site 11: Atemengue pond Obili). At the end of the floristic survey, the selection of the pollutant-tolerant plant species was based firstly on their characteristics as bioaccumulator and their classification as major, intermediate and minor heavy metal accumulator groups. Therefore, three most frequent and abundant plant species were selected according to the above listed criteria. They were *Echinochloa pyramidalis*, *Commelina benghalensis* and *Pennisetum purpureum* and were used to evaluate their capacities in bioaccumulation of heavy metals.

II.2.2.2. Soils, waters, plants sampling and physico-chemical parameters analyses methods

II.2.2.2.1. Soils sampling and physico-chemicals analysis

II.2.2.2.1.1. Soils sampling

Soils were sampled using the auger during dry and rainy season at the same sites and location points as water and plants. According to the soil distribution, soil samples were collected (0-20 cm) from three stations realized within the watershed. P1, P2 and P3 were located upstream, in the middle and downstream respectively. Each sampling point was geo-referenced in order to locate the sites and to set up future field surveys. To avoid contamination, soil samples were taken from the bottom to the top. In order to get composite samples (Hussain et *al*., 2010), soil collected in point P1, P2 and P3 were mixed. One composite sample of equal quantity of soil, two kilogram (2 kg) weight was formed (Mahmood et *al*., 2010; Mandre, 2014), labelled and placed in the plastic container for analysis. At the end of the survey, three composite soils samples were collected from the three sites (Appendix 2). No metal containers were used in order to avoid metal contamination. All the soil samples collected were stored in new plastic bags, labelled and transported to the laboratory for analysis.

II.2.2.2.1.2. Soils physico-chemicals analysis

II.2.2.2.1.2.1. Determination of Organic matter (OM), Organic Carbon (OC), particle size distribution and cation exchange capacity (CEC) in the soil

Organic carbon was determined by chromic acid digestion of 500 mg of finely ground fraction with heating and spectrophotometric analysis (Heanes, 1984) expressed as percentage (%). Organic matter obtained using the formula: OM (%) = OC (%) × 2. Particle size distribution and cation exchange capacity determined using ammonium acetate method at pH 7 and quantified by flame atomic absorption spectrophotometric (AAS) analysis. Particle size (three fractions) was determined by the hydrometer method (Bouyoucos, 1951). Soil texture was measured by Master sizer 2000 (Malvern Instrument, Malvern, the UK) and determined using the USDA soil texture triangle (Groenendyk et *al*., 2015).

II.2.2.2.1.2.2. Determination of the pH H_2O and pH KCl

The pH and electrical conductivity (EC) of soil samples were measured using glass electrode in distilled water (pH H_2O) and 1 $mol.dm^{-3}$ of KCl solution (pH KCl) at a ratio 1:2.5 soil-solution (Kabała et *al*., 2016) using the glass electrode. 10 g of soil, previously dried, ground, sieved (ø 2 mm sieve), and thoroughly mixed, were placed in 250 ml beakers with 25 ml of distilled water and 1 $mol.dm^{-3}$ KCl were added, respectively. After hand mixing, the suspensions were left for 1 hour before pH measurement. The measurements were take in triplicate using a glass electrode in a soil-water suspension after calibration based on standard solutions in a pH range of 4.01-7.01-10.01 (WTW Instruments).

II.2.2.2.2. Water sampling and physico-chemical parameters analysis

Water was sampled using the polyethylene plastics bottles of 1L capacity (previously labelled and rinsed with distilled water) as described by (Rodier, 2009) at the same sampling points as soils. At each site, the bottles were previously rinsed with water to be taken from three different points in the river (upstream, middle and downstream) and the physico-chemical parameters were taken at each of these points (Appendix 2). One composite sample was formed for each site (Kinzelman et *al*., 2006; Cornman et *al*., 2018). Salinity (Sal in ‰), potential hydrogen (pH in units), potential redox (Eh in mV), temperature (T°C in degree) and conductivity (Cnd in µS/cm) were directly measured in the field during the dry and rainy seasons using a WTW pH / Cond / Oxi Multi 3320 SET 1 meter kit. At the end of the sampling, 03 composite water samples were collected during each season for this study. The three water samples were stored in a cooler (4°C), transported to the laboratory, and kept in a refrigerator for heavy metals analysis.

II.2.2.2.3. Plants sampling

In each site, plant samples were also collected. Each plant species of *E. pyramidalis*, *C. benghalensis* and *P. purpureum* were sampled. Leaves and stems were gently separated from roots of each species, placed in the journal paper, which were previously labeled. One composite sample of 100g of roots, stems and leaves was formed for each plant specie sample for each site. The leaves, stems and roots of the three species were washed with water to remove ground attached to the surface. 27 samples (leaves, stems and roots) of the three sites were washed and rinsed with deionized water, oven dried at 60°C for 48 hours and ground to a fine powder. After that, plants were ground to powder for metals analysis.

II.2.2.3. Analysis of heavy metals concentrations in the soil, water and plants

The heavy metals concentrations in all water, soil and plants samples were analyzed by Inductively Coupled Plasma - Optical Emission Spectroscopy (ICP-OES) at the laboratory of the International Institute of Transforming Agriculture (IITA) at Nkolbisson Yaounde-Cameroon.

Soils were air-dried and ground to pass through a 2 mm sieve to remove the coarser soil fraction (Makoi & Verplancke, 2010) and further finely ground to pass through a 0.5 mm sieve. 500 mg of soil sample was digested in aquaregia (Chen & Ma, 2001) and Cd, Cr, Co, Cu, Ni, Pb, As and Zn were analyzed by ICP-OES after calibrating the instrument with certified standards. Results obtained were reported in micrograms per gram (µg/g) (Appendix 3).

1.2 mL of water sample was mineralized in nitric acid 2.8 mL solution in aquaregia and Cd, Cr, Co, Cu, Ni, Pb, As and Zn were analyzed by ICP-OES after calibrating the instrument with certified standards. Results obtained were reported in milligrams per liter (mg/L)

Leaves, stems and roots of the plants were washed and rinsed with deionized water, oven dried at 600 °C for 48 hours and ground to a fine powder for metal analysis (Jones et *al.*, 1990). Trace elements Cd, Cr, Co, Cu, Ni, Pb and Zn were determined by the following procedure: 500mg of finely ground sample was digested in nitric acid, then diluted to 50 ml and let stand overnight. The supernatant was carefully transfer into centrifuge tubes for ICP determinations. The certified standards for requested elements were prepared and after calibrating ICP-OES, Cd, Cr, Co, Cu, Ni, Pb and Zn were determined. Wavelengths (λ) of the ICP for the analyzed metals were: 228.802 nm for Cd; 267.716 nm for Cr; 228.616 nm for Co; 327.393 nm for Cu; 231.604 nm for Ni; 220.353 nm for Pb; and 213.9 nm for Zn (Appendix 4).

II.2.2.3.1. Assessment of heavy metals contamination levels of soils

To estimate the anthropogenic and natural impacts of heavy metal on soil, a common approach was to evaluate the background concentrations levels (Islam et *al.*, 2015; Yi et *al.*, 2020). Unfortunely in Cameroon, and especially in Yaounde, there was no available information about such data, which can be used to assess heavy metal background. However, in their study on the assessment of heavy metals in soils and groundwater in an urban watershed of Yaounde (Cameroon-West Africa), Defo et *al*. (2015) determined the background concentrations of some heavy metal in soil surface in Yaounde (0 - 20 cm) such as Pb, Cd, Cr, Ni in Yaounde-Cameroon. The control values of metals in the soils (background concentrations) related to the geochemistry of the parent materials was calculated through the topsoil samples in rural area of Yaounde far removed from the influence of human activities, without pollution at the outskirts (Defo et *al*., 2015). Nevertheless, this author had determined not all background data. In the current study, the background concentration of Zn and Cu in Yaounde was calculated base on the reference data of other regions or countries. Logan et *al*. (1983) said that specific information of the study area is necessary; otherwise, data published for other regions concerning the background metal levels can be helpful.

II.2.2.3.1.1. Determination of geo-accumulation index (Igeo)

The geo-accumulation index (Igeo) was used to assess the contamination levels of heavy metals and their sources in urban soils. The Igeo was used to determine the heavy metal contamination of soils by comparing current and background concentrations. Background concentrations were used as control. The geo-accumulation index (Igeo) was calculated using the following equation (Ji et *al*., 2008):

$$\boxed{Igeo = \frac{log2ci}{1.5 \times Bn}} \quad (1)$$

where Ci is the concentration of the element in environment, Bn is the geochemical background value in soil. The content of a given substance in the environment used the constant 1.5 to detect a very small anthropogenic influence and to analyze the natural fluctuations (Biggan & Linsheng, 2010). The Igeo for each metal is calculated and classified as follows:

- Igeo ≤ 0 means uncontaminated
- 0 < Igeo ≤ 1 means uncontaminated to moderately contaminated
- 1 < Igeo ≤ 2 means moderately contaminated
- 2 < Igeo ≤ 3 means moderately to heavily contaminated
- 3 < Igeo ≤ 4 means heavily contaminated
- 4 < Igeo ≤ 5 means heavily to extremely contaminated and
- Igeo ≥ 5 means extremely contaminated.

II.2.2.3.1.2. Determination of the spatial distribution of heavy metal in soils

II.2.2.3.1.2.1. Pollution index (PI)

The single pollution index (PI) or contamination factor was used to determine the pollution of the metal in the soil. This index helped to assess if the soils were polluted by the selected metal or not. The pollution index of each element i was defined as the ratio of the metal concentration (Ci) in the sampling point to the background concentration (Bn) of the corresponding metal as the following formulation (Ji et *al*., 2008).

$$\boxed{PIi = \frac{Ci}{Bn}} \qquad (2)$$

where Ci is the measured concentration of an element i in the environment and Bn is the geochemical background value in soil. PIi ≤ 1 means that the soil is not polluted; PIi > 1 means that the soil is polluted.

II.2.2.3.1.2.2. Nemerow integrated pollution index (IPI)

The Nemerow integrated pollution index (IPI) was used to perform the pollution level of soils. The Nemerow IPI considers not only the mean values of pollution index of all considered metals (averPIi) but also their maximum value (maxPIi). The Nemerow IPI was computed by the following formula (Lu et *al*., 2014; Yang et *al*., 2014):

$$\boxed{IPI = \sqrt{\frac{aver(Pli)2+max(Pli)2}{2}}} \qquad (3)$$

where max (PIi) is the maximum value of the single pollution indices of all heavy metals and aver(PIi) is the average value of the pollution index of heavy metals at a given sampling point. IPI is useful for classifying soil pollution level. The IPI is classified as follows: IPI ≤ 0.7, safe; 0.7 < IPI ≤ 1.0, precaution; 1.0 < IPI ≤ 2.0, slight pollution; 2.0 < IPI ≤ 3.0, moderate pollution; and IPI ≥ 3.0, heavy pollution (Cheng et *al*., 2014).

II.2.2.3.1.3. Assessment of the potential ecological risk index (RI)

Soil ecological risk index (RI) evaluation considered the toxicity level, the synergistic effect, and ecological sensitivity of various potentially toxic metal elements. The calculation formula is showed by equation (4):

$$\boxed{RI = \sum E_r^i = \sum T_r^i \times CF_i = \sum T_r^i \times \left(\frac{C_s^i}{C_b^i}\right)} \qquad (4)$$

where T^i_r is the toxicity response coefficient. The toxicity response coefficient of Cu, Cd, Pb, Cr, Ni, As, Zn and Co were 5, 30, 5, 2, 5, 10, 1 and 5, respectively (Xiang et *al*., 2019; Tan & Aslan, 2019). E^i_r is the potential ecological risk index. Five levels were recognized: $E^i_r < 40$, slight risk; $40 \le E^i_r < 80$, moderate risk; $80 \le E^i_r < 160$, high risk; $160 \le E^i_r < 320$ very high risk; $E^i_r \ge 320$, extremely high risk.

RI is the comprehensive potential ecological risk index of various metals in soil, which consists of four classes: RI < 150, slight risk; 150 ≤RI< 300 moderate risk; 300 ≤RI< 600, high risk; RI≥ 600 very high risk.

II.2.2.3.2. Assessment of the level of water pollution by heavy metals

II.2.2.3.2.1. Heavy metal toxicity load (HMTL)

The heavy metal toxic load (HMTL) indicates the content of metals in the water body. HMTL is a factor that evaluates the toxic metals level found in water. It provides the quantity of heavy metal present in the water that may affect living body (human health or animal). It gives an idea to the regulatory authority about the extent of treatment required to treat the lowland water to acceptable levels for human use purposes. This technique can help to assess and provide an effective treatment and management plan for water in lowland areas. HMTL was evaluated by multiplying the measured concentration of heavy metals with its hazard intensity as presents below:

$$\boxed{HMTL = \sum_{i=1}^{n} C \times HIS} \qquad (1)$$

where C is the concentration of heavy metal; n is the number of heavy metals and HIS is the hazard intensity score which is obtained from the ATSDR (Anonymous 7, 2019; Proshad et *al*., 2020). HIS is allocated based on the frequency of incidence of toxic metals as a harmful substance on the National Priorities List (NPL) sites maintained by ATSDR, the toxicity level of studied metals, and the prospect of human contact. The maximum HIS for toxic metal is 1800, where 600 points are allotted for each of the NPL frequency, the toxicity, and the prospect of human contact.

The percentage of metal removal from water to reduce the pollutant load was estimated by the following formula (Proshad et *al*., 2020):

$$\boxed{\text{TMRPL (\%)} = \{\frac{[THMTL-PTL]}{[tHMTL]}\} \times 100} \qquad (2)$$

where THMTL is the total of the heavy metals toxicity load at all sampling sites; PTL is the allowable toxicity load and tHMTL is the total of the heavy metals toxicity load that was calculated.

II.2.2.3.2.2. Heavy metal evaluation index (HEI)

The heavy metal evaluation index (HEI) represents an overall quality of the water with respect to heavy metals in water samples (Edet & Offiong, 2002; Taiwo et *al*., 2020) and was calculated as follows:

$$\boxed{HEI = \sum_{i=1}^{n} Hc/Hmac} \qquad (3)$$

where Hc is the monitored value of the i^{th} parameter and Hmac is the maximum admissible concentration of the i^{th} parameter. The maximum admissible concentration for Pb, Cd, Cr, Ni, Cu, As, Zn and Co were 0.05, 0.005, 0.05, 0.02, 0.05, 0.04, 1 and 5 mg/L, respectively (Anonymous 8, 2017). The HEI is classified as follows: low (< 10), medium (10 <HEI< 20), and high (> 20).

II.2.2.3.2.3. Ecological risk assessment

The ecological risk index (ERI) of water was calculated using the functions described in equations (3 and 4) (Taiwo et *al*., 2020):

$$\boxed{ERI = \sum RI = \sum Ti \times PI} \qquad (4)$$

$$\boxed{PI = Cs/Cb} \qquad (5)$$

where RI is the potential ecological risk factor of each heavy metal; Ti is the toxic response factor of heavy metal; PI is the pollution index; Cs is the concentration of heavy metals in the sample; and Cb is the corresponding background values. The toxic response factor of each studied trace metals are 5, 30, 1, 5, 5, 10, 1 and 5 for Pb, Cd, Cr, Ni, Cu, As, Zn and Co, respectively (Taiwo et *al*., 2020). ERI value <150 indicates low ecological risk, 150<ERI<300 indicates moderate ecological risk, 300<RI<600 indicates considerable ecological risk, and ERI>600 indicates very high ecological risk (Taiwo et *al*., 2020).

II.2.3. Metal bioaccumualation efficiency

II.2.3.1. Determination of plants accumulation capacities

To evaluate the capacities of plants to metal accumulation, the following parameters were used: translocation factor (TF), mobility ratio (MR), and bioaccumulation factor (BAF) (Kandziora-Ciupa et *al*., 2017).

II.2.3.1.1. Translocation factor (TF)

The translocation factor is the plant capacity to translocate metal from the roots to the shoots. TF is the ratio of metal concentration in the shoots (leaves + stems) to the roots. It was determined by the formula:

$$\boxed{TF = \frac{[Metal]shoots}{[Metal]roots}} \qquad (1)$$

- TF greater than 1 (>1), means that the plant effectively translocate heavy metals from roots to the shoots.
- TF less than 1 (<1), indicates that metals accumulated by plants are largely retained in the roots (Rezvani & Zaefarian, 2011).

II.2.3.1.2. Mobility ratio (MR)

MR is the ratio of metal concentration in the shoots (leaves + stems) to its concentration in the soil.

- MR>1 indicates that the plant is enriched with metals (accumulator);
- MR=1 indicates a rather indifferent behavior of the plant towards metals (indicator);
- MR<1 shows that the plant excludes metals from uptake (excluder) (Mingorance et *al*., 2007, Serbula et *al*., 2012).

$$\boxed{MR = \frac{[Metal]shoots}{[Metal]soil}} \qquad (2)$$

II.2.3.1.3. Bioaccumulation factors (BAF)

Bioaccumulation or bioconcentration Factor (BAF or BCF) was used to quantify the accumulation efficiency of toxic elements in plants. It was used to measure the ability of each organ (leaves, stems and roots) to accumulate metals from the soil (Hladun et *al*., 2015), comparing the concentration in the plant parts and an external environment (Rezvani & Zaefarian, 2011). BAF>1 indicates that particular element is accumulated by leaves, stems, or roots from the soil (Yoon et *al*., 2006; Serbula et *al*., 2013).

BAF was categorized as follows:

- BAF<1 indicates that plant is excluder;
- 1<BAF<10 indicates that plant is accumulator; and
- BAF>10 indicates that plant is hyperaccumulator (Jha et *al*., 2016).

$$\boxed{BAF = \frac{[Metal]plants\ tissues}{[Metal]soil}} \qquad (3).$$

II.2.3.2. Assessment of plant accumulation performances

II.2.3.2.1. Metal accumulation index (MAI)

The metal accumulation index (MAI) was assessed to give the overall performance of heavy metal accumulation in plants species. It was used according to the following calculation:

$$\boxed{MAI = \left(\frac{1}{N}\right) \sum_{j=1}^{N} Ij} \qquad (4)$$

$$\boxed{Ij = \frac{x}{\Delta x}} \qquad (5)$$

where N is the total number of metals analyzed, Ij is the sub-index for variable j, obtained by dividing the mean value (x) of each metal by its standard deviation Δx (Liu et *al*., 2007; Hu et *al*., 2014).

II.2.3.2.2. Comprehensive bioconcentration index

The Comprehensive bioconcentration index (CBCI) is the assessment of plant species based on their ability to accumulate multiple metals in their system. The following steps were considered for the calculation of the CBCI as proposed by Zhao et *al*. (2014). At the beginning, the fuzzy set/factor set (U) was established as follows:

$$\boxed{U = (u_1, u_2, u_3, \ldots u_i)} \qquad (6)$$

where U indicated the comprehensive accumulation capability of plant species, and u_i corresponds to the different heavy metal influence factors (Pb, Cd, Cu, Zn, Cr, Ni, As and Co).

Secondly, the fuzzy membership function was estimated as follows:

$$\boxed{\mu(x) = \frac{x - x_{min}}{x_{max} - x_{min}}} \qquad (7)$$

where x is the BAF of a specific metal. Minimum and maximum BAF values were represented by x min and x max for the given metal among the observed plant species. The fuzzy membership quotient, μ(x) ranges between 1 and 0 signifying the highest and lowest comprehensive accumulation potential of plant species to different metals.

Lastly, CBCI was evaluated using equation (2):

$$\boxed{CBCI = \left(\frac{1}{N}\right) \sum_{j=1}^{N} \mu_i} \qquad (8)$$

where N = total number of metals analyzed and μ_i = μ(x) of metal i.

II.2.4. Data analysis

All statistical analyses were performed using R (3.4.1) and R-studio software. The Biodiversity "R" package integrated into R (3.4.1) was used to perform statistical analyses on the floristic data. QGIS 2.18.x was used for mapping map the geospatial distribution of macrophyte species in the selected lowland site and potential pollution sources on a 1:30,000 scale base map of Yaounde (WRI and MINFOF, 2013; Open Street Map, 2018). R library packages (Factor MineR, Ggplot2) were used to perform multivariate statistical analysis such as PCA, Cluster and correlation analysis tests to explore groups and sets of heavy metal analyses variable in soils, water and plants with similar properties (Wang et *al*., 2017; Gao et *al*., 2020). IBM SPSS software (version 20.0 for Windows) was used to perform the ANOVA between the different parameters analyzed. The map of the sampling sites was generated from Google Maps using Global Positioning System (GPS) coordinates.

II.2.4.1. Floristic survey data

In order to assess significant differences between sites for all macrophyte species, the Shapiro-Wilks test was used to see if the data obtained follow a normal distribution or not. For data following a normal distribution, parametric tests (one-way analysis of variance (ANOVA) at the 5% significance level and turkeys) were used. Then, for data not following the normal distribution, non-parametric tests (Kruskals-Wallis and Wilcoxon) were used. Multivariate statistical methods using cluster analysis (CA) and principal component analysis (PCA) was performed for the spatiotemporal distribution of macrophytes (Achi et *al*., 2021). CA explores clusters and sets of variables with similar properties, potentially simplifying the description of observations by finding structure or patterns in the presence of chaotic or confusing data. Floristic diversity for the distribution of major macrophyte groups in relation to lowland sites was established.

II.2.4.2. Level of contamination of soil, water and plants by heavy metals

Data obtained from the analytical methods were processed statistically using IBM SPSS software (version 20.0 for Windows). A one-way ANOVA was performed with a 95% confidence interval, with statistical significance defined as $p<0.05$, for describing the temporal variations of the observed water quality parameters and to determine significant differences between water sampling seasons (rainy and dry) for the different metal contents. Two-way analysis of variance (ANOVA) was used to determine differences and interactions between sites (4, 9, and 11) and the other matrices (soil, sediment, and water). Indeed, before studying the site and season effects on the different parameters analyzed during both seasons, two-way ANOVA was used to test for differences in data between sites, species, seasons, and plant tissues, and between plants tissues x plant species for each site (4, 9 and 11) and seasons (rainy and dry). For data, which did not follow the normal distribution, non-parametric tests (Duncan and Kruskals-Wallis) were used.

Numerical analysis was performed using Microsoft Excel to calculate the mean, standard deviation, geoaccumulation index (Igeo), soil ecological risk (ERI), and heavy metal toxic load (HMTL), heavy metal assessment index (HEI); ecological risk (ERI) in water. Pearson's correlation (r) was used to show the degree of association between the studied parameters and plant part values. One-way analysis between groups ANOVA was performed to compare the effect of rainy and dry seasons on different physicochemical properties of water and soil samples from different sites.

In addition, multivariate analysis using principal component analysis (PCA) and cluster analysis (CA) was used to identify potential related and likely sources of pollution, both natural and anthropogenic (Wang et *al*., 2017; Gao et *al*., 2020). In PCA, the principal components were calculated based on the correlation matrix. Cluster analysis (CA) was performed to rank heavy metals from different sources based on similarities in their chemical properties. Pearson's correlation coefficient was used to measure the degree of correlation between physicochemical parameters of soils, water and metal concentrations in soils, water and plants and to identify the relationship between pairs of elements.

II.2.4.3. Remediation performance of plant species in metal accumulation

Data were calculated using Microsoft excel to assess the capabilities of plants in metal accumulation, factors such as TF, MR, BAF and Metal Accumulation Index (MAI) in water and the Global Bioconcentration Index (CBCI) of plants, and analyses were using IBM SPSS software to calculate the mean, standard deviation. Pearson correlation analysis was performed to determine the interaction between heavy metals and plants in both compartments (water and soil). The correlation between each variable was quantified by performing a Pearson analysis and testing the significance of the r-values at the p-levels equal to 0.05 and 0.01. The R libraries FactoMineR and Hmisc were used to perform PCA and correlation tests, respectively. PCA was applied to the resulting multivariate data of plant root and shoot CBCI values for the seven heavy metals (Pb, Cd, Cr, Ni, Zn, Cu, As, and Co) at sites 4, 9, and 11.

CHAPTER III. RESULTS AND DISCUSSION

III.1. Results

III.1.1. Floristic diversity of lowlands in Yaounde

III.1.1.1. Taxonomic richness of families

During the rainy season, the results presents that, the most diversified families identified in the polluted sites were Poaceae (23 species), Asteraceae (20 species), Fabaceae (14 species), Malvaceae (12 species) and Solanaceae (12 species). Less than 10 species belonged to other families. For polluted sites, families represented by just one specie each were Annonaceae, Apiaceae, Apocynaceae, Atyraceae, Burseraceae, Cannaceae, Caricaceae, Caryophyliaceae, Commelinaceae, Labiateae, Lamiaceae, Lauraceae, Lemnaceae, Moringaceae, Nyctaginaceae, Passifloraceae, Piperaceae, Portulacaceae, Strelitziaceae, Ulmaceae, Verbenaceae and Vitaceae (Fig. 15).

Compared to the dry season, the family of Poaceae (19 species) was the most widespread. It was followed in the descending order by the families of Asteraceae (17 species), Cyperaceae (9 species), Convolvulaceae (8 species), Fabaceae (8 species). As for the other families, less than 7 species were recorded. The families of Apiaceae, Atyraceae, Bromeliaceae, Cannaceae, Caricaceae, Costaceae, Dioscoreaceae, Hydroleaceae, Lauraceae, Marantaceae, Nyctaginaceae, Oxalidaceae and Piperaceae were represented by only one specie (Fig. 16).

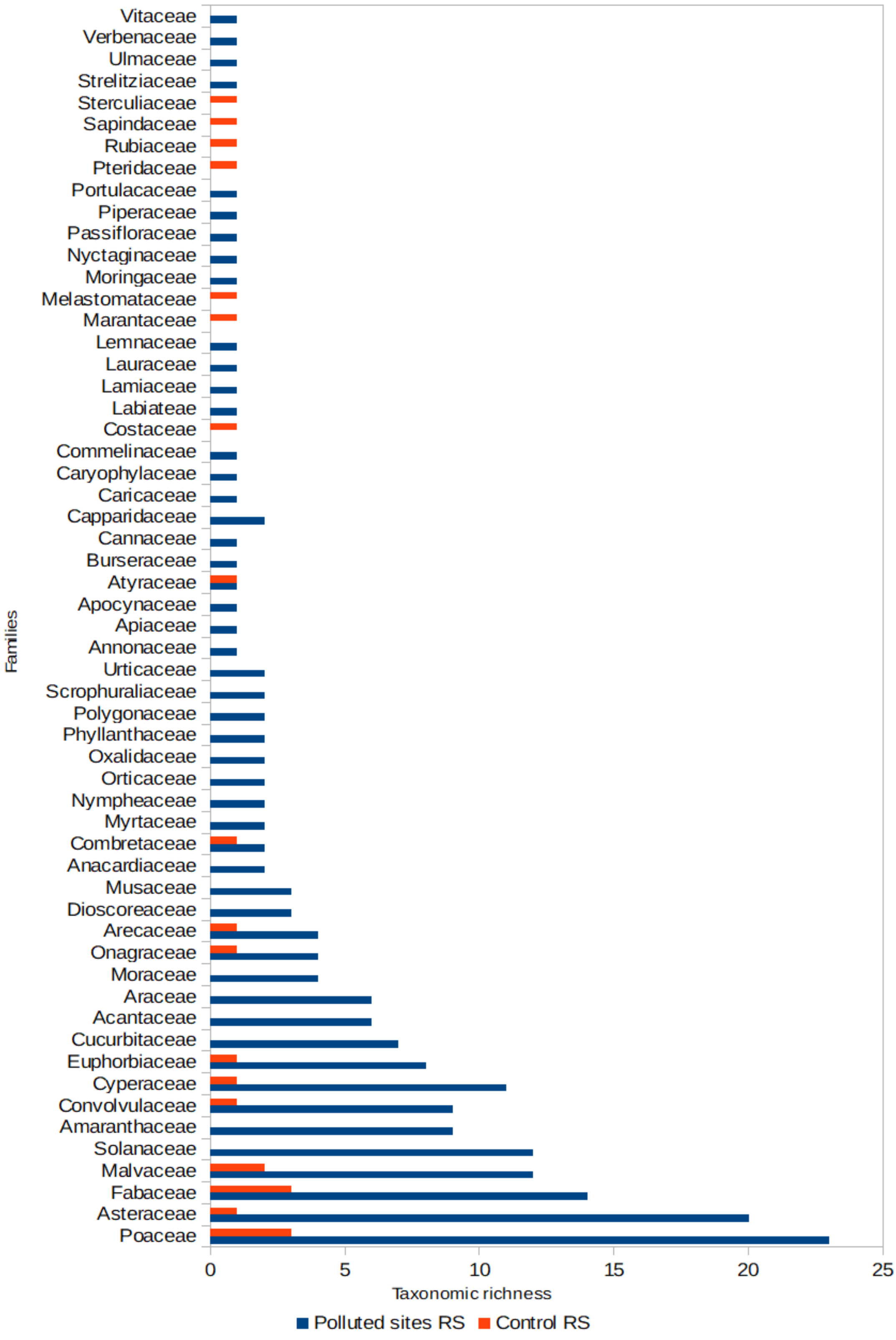

Fig. 15. Taxonomic richness of families growing on polluted and control sites during the rainy season. (RS: rainy season).

In the control site during both seasons, the families represented were Combretaceae, Marantaceae, Melastomataceae, Pteridaceae, Rubiaceae, Sapindaceae, Sterculiaceae, Bromeliaceae and Hydroleaceae. These families were not found in polluted sites. Poaceae was the well represented family and it was reported to preferentially accumulate lead (Pb) and cadmium (Cd).

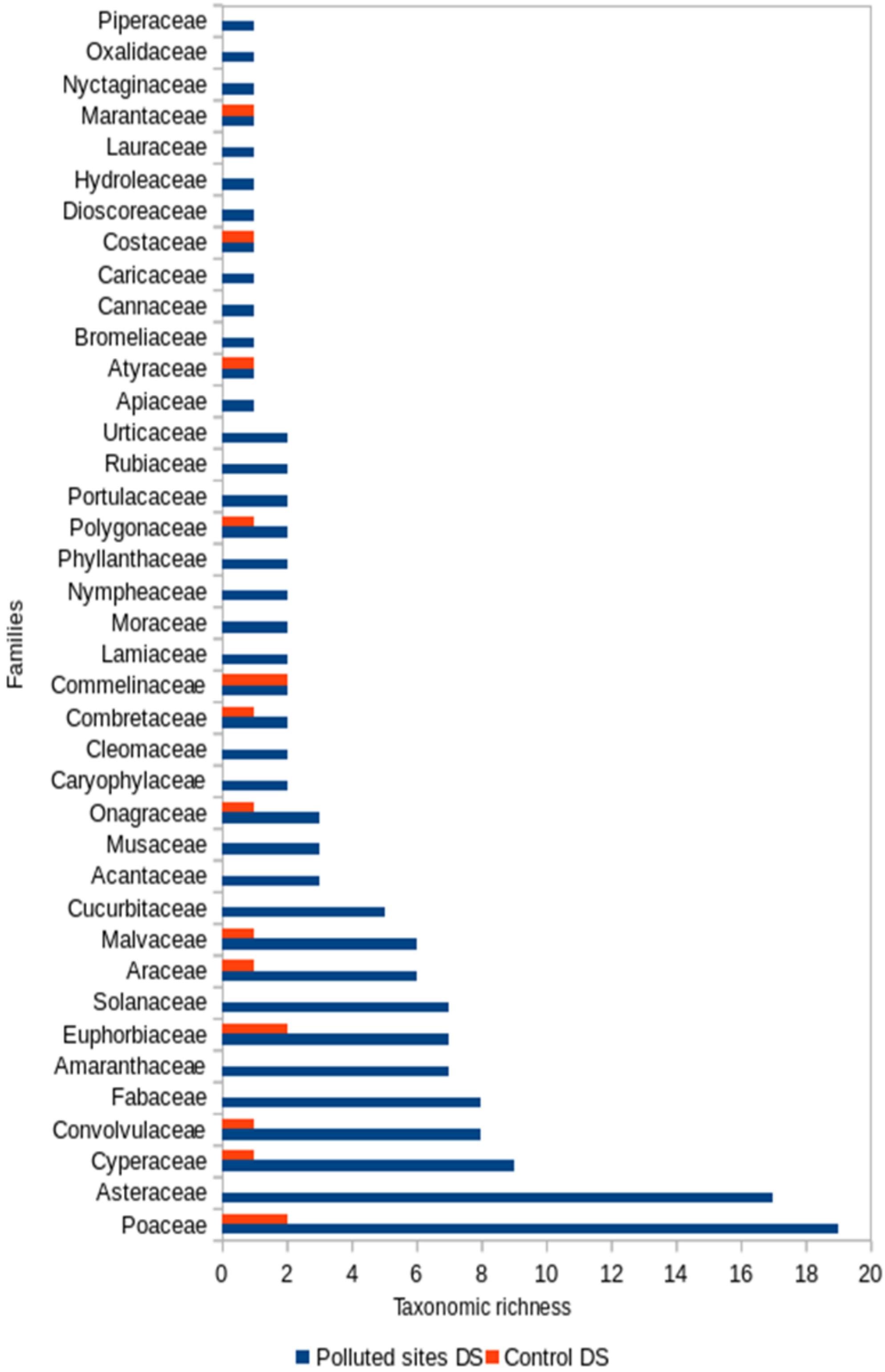

Fig. 16. Taxonomic richness of families growing on polluted and control sites during the dry season. (DS: dry season).

III.1.1.2. Species diversity

Species richness (S) varied from 22 to 77 in polluted sites compared to 23 in the control site during the rainy season. The Shannon diversity index (H') of polluted sites ranged from 1.51 to 3.68 bits/ind. compared to 2.34 bits/ind. in control (Fig. 17a & b). Considering Species richness (S) and Shannon index (H'), and ANOVA test respectively (Pr<p) =1.7e-10 and (Pr<p) =2.29e-7 showed a significant difference between the sites at the probability threshold of 5%. This reflects the highest species variation between all the sites. Pielou equitability index (J') in the polluted sites ranges from 0.459 to 0.847 compared to the control with 0.747. There was no significant difference (Pr>p =0.0688) according to the Kruskals-Wallis test. Pielou's equitability index (J') values and the Simpson's index gave evidence of an almost equitable diversity in the abundance of most species whith bioaccumulators characteristics (Fig. 17c). This was despite the fact that in each site there was a dominance of the floristic background by some species. Therefore, all these values, close to the maximum value (J'=1) reflected an equitable distribution of individuals within species at all the sites. The results were similar for each site (Table VI). The high Shannon diversity index values at polluted sites may reflect a dominant trend of greater floristic diversity. Sorensen index value obtained by comparing all polluted sites and the control was much lower than 50% (K=7.9%) indicated that the plant species communities were different and that there was no similarity in the flora between the two types of sites investigated. This trend was also observed between each polluted site and the control where K<12% except in site 3 (Nkolbisson lake) where K reached 21.4%.

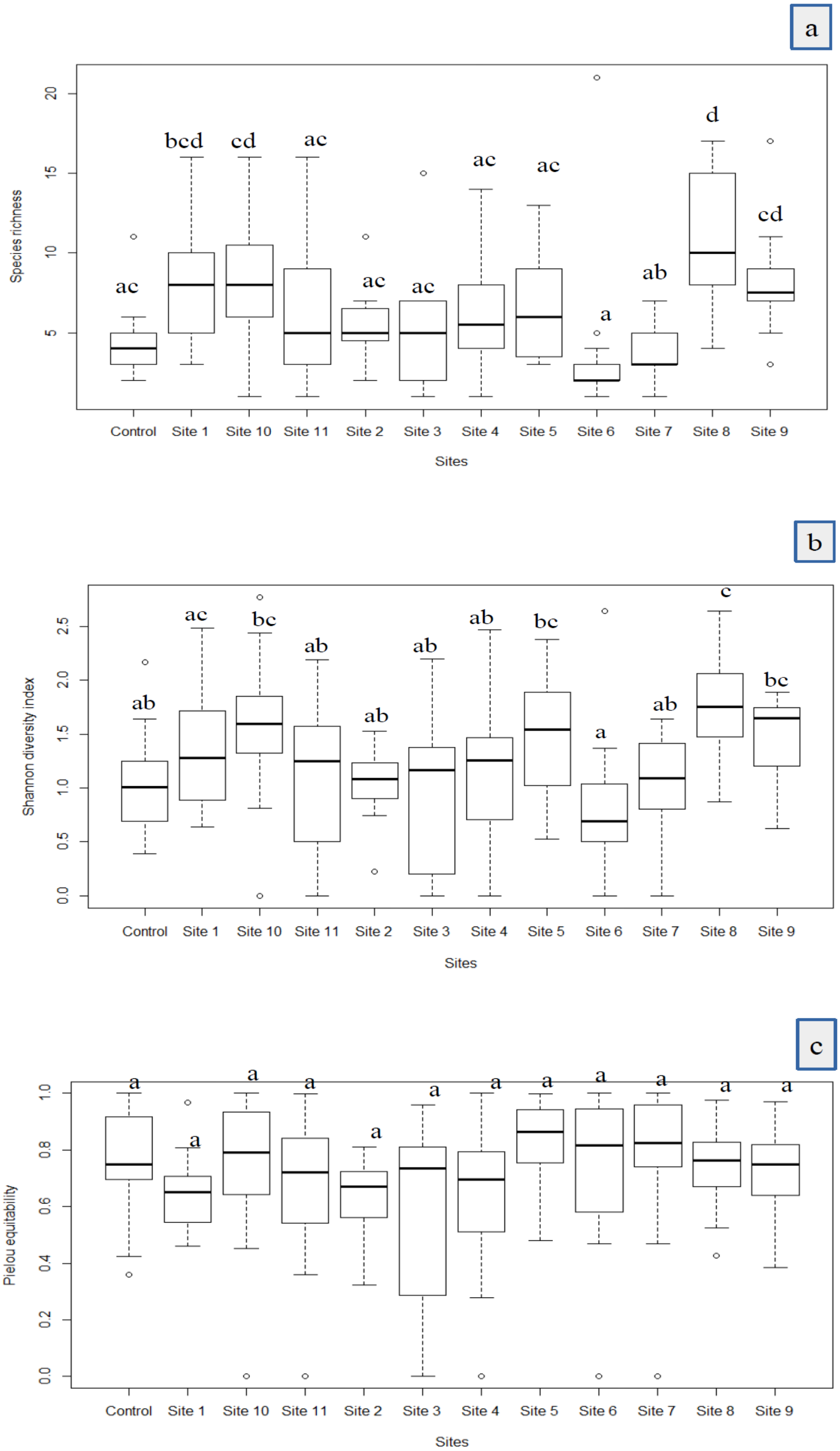

Fig. 17. Change in species richness (a), Shannon diversity index (b) and the Pielou equitability index (c) in polluted lowland sites compared to the control boxes during the rainy season. ([a] and [b, c] statistically not significant and significant at $p<0.05$).

Table VI. Taxonomic richness and diversity indices of macrophytes in control and lowland polluted sites surveyed in Yaounde during the rainy season. (Mean values followed by the same letter are not significantly different at $p < 0.05$).

Lowland sites	Floristic composition			Diversity index			
	Family richness	Genera richness	Species richness (S)	Shannon index (H')	Pielou equitability index (J')	Simpson index (D)	Sorensen index (K)
Site 1 : Cite-verte	16	35	42 ± 4^{bcd}	2.90 ± 0.55^{ac}	0.775 ± 0.14^{a}	0.912	6.2
Site 2 : Nkolbisson (Nkol-da)	13	24	29 ± 2^{ac}	2.13 ± 0.33^{ab}	0.631 ± 0.14^{a}	0.791	3.9
Site 3 : Nkolbisson (IRAD	20	32	33 ± 4^{ac}	2.82 ± 0.72^{ab}	0.806 ± 0.35^{a}	0.914	21.4
Site 4 : Mokolo-elobie	23	44	55 ± 3^{ac}	3.02 ± 0.54^{ab}	0.753 ± 0.22^{a}	0.921	10.3
Site 5 : Messa	21	38	45 ± 3^{ac}	3.02 ± 0.2^{bc}	0.795 ± 0.13^{a}	0.928	11.8
Site 6 : Municipal lake	16	27	27 ± 4^{a}	1.51 ± 0.57^{a}	0.459 ± 0.29^{a}	0.645	8.0
Site 7 : Retenue pond (UYI)	12	19	22 ± 2^{ab}	2.49 ± 0.48^{ab}	0.805 ± 0.27^{a}	0.883	4.4
Site 8 : Ngousso	27	57	70 ± 4^{d}	3.32 ± 0.47^{c}	0.782 ± 0.12^{a}	0.950	6.5
Site 9 : Mvan	18	45	55 ± 3^{cd}	3.10 ± 0.39^{bc}	0.775 ± 0.13^{a}	0.930	5.1
Site 10 : Biyem-assi	36	70	77 ± 3^{cd}	3.68 ± 0.60^{bc}	0.847 ± 0.24^{a}	0.959	8.0
Site 11: Atemengue pond (Obili)	25	46	54 ± 4^{ac}	2.63 ± 0.63^{ab}	0.659 ± 0.24^{a}	0.860	10.4
Site 12: Control (Ongol village)	18	23	23 ± 2^{ac}	2.34 ± 0.47^{ab}	0.747 ± 0.19^{a}	0.859	
Study zone	60	138	189	3.89	0.742	0.957	
P-values			$1.7e^{-}10$	$2.29e^{-}7$	0.0688		

Compared to the dry season, the diversity index respectively ranged from 13 to 50 for Species richness (S), 1.88 to 3.16 bits/ind. for Shannon diversity index (H') and 0.692 to 0.819 for Pielou's equitability index (J') in polluted sites than 17, 2.45 bits/ind. and 0.866 in control (Fig. 18a, b and c). Considering Species richness (S) (Pr<p) =0.00052, Shannon and Weaver index (H') (Pr<p) =0.00242 and Pielou's equitability index (J') (Pr<p) =0.00447, ANOVA test showed a significant difference between all sites at a 5% probability level. This reflects the fact that sites were diversified and a maximum of species participated in the recovery of the areas. Pielou's equitability and Simpson index values close to the maximum value (J'=1) showed an equitable distribution of individuals within species at all sites (Fig. 18c). This trend was also observed at each site (Table VII).

Table VII. Taxonomic richness and diversity indices of macrophytes in control and lowland polluted sites surveyed in Yaounde during the dry season. (Site 12: Control, Mean values of the diversity index followed by the same letter are not significantly different at $p < 0.05$).

Lowland sites	Floristic composition		Diversity index				
	Family richness	Genera richness	Species richness (S)	Shannon index (H')	Pielou equitability index (J')	Simpson index (D)	Sorensen index (K)
Site 1	14	24	$27±3^{ac}$	$2.66±0.59^{ab}$	$0.808±0.150^{b}$	0.903±0.239	14
Site 2	12	26	$32±3^{c}$	$2.51±0.36^{b}$	$0.724±0.103^{ab}$	0.877±0.106	12
Site 3	21	40	$44±3^{ac}$	$2.88±0.68^{ab}$	$0.761±0.212^{ab}$	0.904±0.294	23
Site 4	17	30	$34±2^{ab}$	$2.44±0.43^{ab}$	$0.692±0.166^{ab}$	0.864±0.211	8
Site 5	14	22	$13±1^{ab}$	$1.88±0.33^{ab}$	$0.732±0.174^{ab}$	0.803±0.207	33
Site 6.	18	25	$38±2^{ac}$	$2.74±0.41^{b}$	$0.754±0.134^{b}$	0.893±0.154	11
Site 7	23	39	$39±3^{ac}$	$3.00±0.42^{ab}$	$0.819±0.180^{ab}$	0.932±0.163	18
Site 8	20	39	$50±2^{bc}$	$3.16±0.39^{b}$	$0.808±0.173^{b}$	0.937±0.157	9
Site 9	15	30	$37±3^{ac}$	$2.81±0.70^{ab}$	$0.777±0.241^{ab}$	0.898±0.294	15
Site 10	16	26	$31±1^{a}$	$2.52±0.29^{a}$	$0.735±0.222^{a}$	0.884±0.170	13
Site 11	12	19	$22±2^{ab}$	$2.29±0.42^{ab}$	$0.741±0.082^{ab}$	0.870±0.151	15
Site 12	13	16	$17±1^{ab}$	$2.45±0.21^{ab}$	$0.866±0.117^{ab}$	0.889±0.102	
Study zone	39	103	139	3.70	0.751	0.950	
P-values			0.00052	0.00242	0.00447	0.00395	

For all polluted sites compared to the control, Sorensen index values were much lower than 50% (K=12.24%). Therefore, between each polluted site and the control, for the majority of sites, K was lower than 23% (K<23) except for site 5 (Messa WWTP) where K was 33%. This means that there was no similarity in the flora between the contaminated and the control sites.

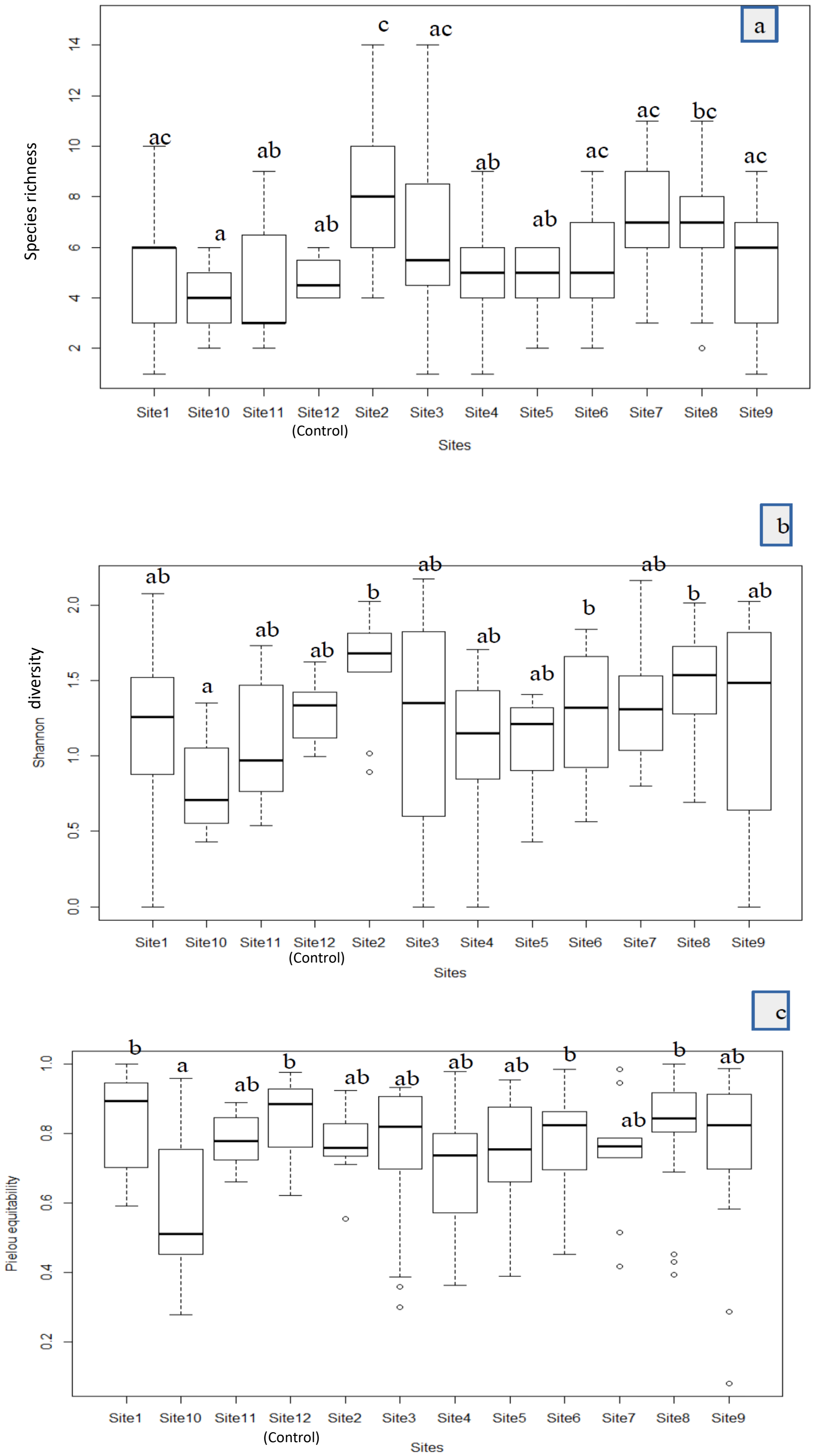

Fig. 18. Change in species richness (a), Shannon diversity index (b) and Pielou equitability index (c) in polluted lowland sites compared to the control boxes during the dry season. ([a] and [b, c] statistically not significant and significant at $p<0.05$).

Compared to the dry season, Sorensen index values during the rainy season was also lower (KRS<KDS<50%). This means that seasons did not influence the flora of bioaccumulators species in terms of similarity between sites.

III.1.1.3. Relative frequency and relative abundance of macrophytes

Ipomoea aquatica had the highest relative frequency (85.7%) in the polluted sites during the rainy season. It was followed by *Echinochloa pyramidalis* (83.9%), *Commelina benghalensis* (75%), *Pennisetum purpureum* (75%), *Panicum maximum* (72.7%), *Ipomoea batatas* (66.7%), *Cynodon dactylon* (65%), *Ageratum conizoides* (60%), *Alternanthera ficoidea* (58.3%), *Setaria barbata* (58.3%) and *Colocassia esculentus* (56.3%). The relative frequency of species varied between the different sites. However, *C. benghalensis* remained the most common macrophyte specie in all polluted sites with some variation (Table VIII).

Compared to the dry season, *E. pyramidalis* (86%) and *C. benghalensis* (86%) were the two species that presented the highest relative frequency. They were followed by *Lycopersicum esculentum* (80%), *Ipomoea batatas* (70%), *Cynodon dactylon* (65%), *Panicum maximum* (64%), *Acalypha hispida* (63%) and *I. aquatica* (55%). In all polluted sites, *E. pyramidalis* was the most common specie (Table VIII).

Concerning the overall plant diversity found in the lowlands of the study area during the rainy season, the most abundant species *were Echinochloa pyramidalis* (12.3%), followed by *Ipomoea aquatica* (8.5%), *Commelina benghalensis* (8.2%), *Pennisetum purpureum* (6.5%), *Setaria barbata* (4.2%), *Panicum maximum* (4.1%), *Ipomoea batatas* (2.9%), *Alternanthera sessilis* (2.2%) and *Alchornea cordifolia* (2%). During the dry season, *E. pyramidalis* (13.9%) was the most abundant specie in all sites, followed by *C. benghalensis* (11.6%), *P. purpureum* (8.1%), *P. maximum* (4.2%), *I. aquatica* (3.9%), *I. batatas* (3.7%), *Alternanthera sessilis* (3.6%), *Nymphea alba* (2.3%), *Polygonum lanigerum* (2.3%), *Ludwigia abyssinica* (2.2%), *Alternanthera ficoidea* (2%) and *Leersia hexandra* (2%). Although these dominant species were almost the same across all sites during both seasons, their rank differed. Plant species abundance in each site during the rainy and dry seasons are presented in Fig. 19, Fig. 20.

In general, during both seasons, the most abundant macrophytes were not the most frequent in the polluted sites also in the control site and vice versa. These results could be used to explain the differences in species diversity observed between polluted and control sites.

Table VIII. Relative frequency of some taxa in the lowland polluted sites (RS: rainy season, DS: dry season).

Sites	RS			DS		
	Families	Macrophytes	Relative frequency (%)	Families	Macrophytes	Relative frequency (%)
Site 1	Amaranthaceae	*Alternanthera ficoidea*	58.3%	Asteraceae	*Acanthospermum hispidum*	50%
	Commelinaceae	*Commelina benghalensis*	58.3%	Amaranthaceae	*Alternanthera ficoidea*	50%
	Poaceae	*Setaria barbata*	58.3%	Commelinaceae	*Commelina benghalensis*	50%
Site 2	Commelinaceae	*Commelina benghalensis*	75%	Commelinaceae	*Commelina benghalensis*	80%
	Poaceae	*Pennisetum purpureum*	75%	Solanaceae	*Lycopersicum esculentum*	80%
	Convolvulaceae	*Ipomoea batatas*	66.7%	Convolvulaceae	*Ipomoea batatas*	70%
Site 3	Commelinaceae	*Commelina benghalensis*	60%	Poaceae	*Echinochloa pyramidalis*	75%
	Onagraceae	*Ludwigia abyssinica*	40.%	Commelinaceae	*Commelina benghalensis*	55%
	Asteraceae	*Acanthospermum hispidum*	20.%	Solanaceae	*Physalis angulata*	35%
Site 4	Commelinaceae	*Commelina benghalensis*	63.2%	Commelinaceae	*Commelina benghalensis*	86%
	Poaceae	*Echinochloa pyramidalis*	47.4%	Poaceae	*Echinochloa pyramidalis*	67%
	Poaceae	*Setaria barbata*	42.1%	Cucurbitaceae	*Cucumeropsis mannii*	43%
Site 5	Araceae	*Colocasia esculenta*	56.3%	Poaceae	*Echinochloa pyramidalis*	86%
	Poaceae	*Echinochloa pyramidalis*	56.3%	Commelinaceae	*Commelina benghalensis*	77%
	Commelinaceae	*Commelina benghalensis*	43.8%	Amaranthaceae	*Alternanthera sessilis*	45%
Site 6	Convolvulaceae	*Ipomoea aquatica*	85.7%	Poaceae	*Echinochloa pyramidalis*	82%
	Poaceae	*Echinochloa pyramidalis*	71.4%	Poaceae	*Panicum maximum*	64%
	Poaceae	*Leersia hexandra*	23.8%	Convolvulaceae	*Ipomoea aquatica*	55%
Site 7	Poaceae	*Echinochloa pyramidalis*	53.9%	Poaceae	*Echinochloa pyramidalis*	40%
	Convolvulaceae	*Ipomoea aquatica*	46.2%	Convolvulaceae	*Ipomoea batatas*	40%
	Fabaceae	*Pueraria phaseoloides*	30.8%	Asteraceae	*Vernonia amygdalina*	40%
Site 8	Commelinaceae	*Commelina benghalensis*	70%	Commelinaceae	*Commelina benghalensis*	65%

	Poaceae	*Cynodon dactylon*	65%	Poaceae	*Cynodon dactylon*	65%
	Asteraceae	*Ageratum conyzoides*	60%	Amaranthaceae	*Alternanthera sessilis*	45%
Site 9	Poaceae	*Panicum maximum*	72.7%	Euphorbiaceae	*Acalypha hispida*	63%
	Amaranthaceae	*Alternanthera sessilis*	54.6%	Asteraceae	*Acanthospermum hispidum*	42%
	Convovulaceae	*Commelina benghalensis*	50%	Asteraceae	*Acmella uliginosa*	37%
Site 10	Commelinaceae	*Commelina benghalensis*	57.9%	Cucurbitaceae	*Zehneria scabra*	44%
	Poaceae	*Pennisetum purpureum*	47.4%	Poaceae	*Pennisetum purpureum*	39%
	Asteraceae	*Ageratum conyzoides*	42.1%	Convolvulaceae	*Ipomoea batatas*	33%
Site 11	Poaceae	*Echinochloa pyramidalis*	83.9%	Poaceae	*Echinochloa pyramidalis*	64%
	Convolvulaceae	*Ipomoea aquatica*	45.2%	Convolvulaceae	*Ipomoea aquatica*	55%
	Commelinaceae	*Commelina benghalensis*	41.9%	Commelinaceae	*Commelina benghalensis*	45%
Site 12	Euphorbiaceae	*Alchornea cordifolia*	78.6%	Atyraceae	*Diplansium sammatii*	75%
Control	Costaceae	*Costus afer*	57.1%	Euphorbiaceae	*Ludwigia abyssinica*	75%
	Poaceae	*Acroceras zizanoides*	42.9%	Onagraceae	*Alchornea cordifolia*	50%

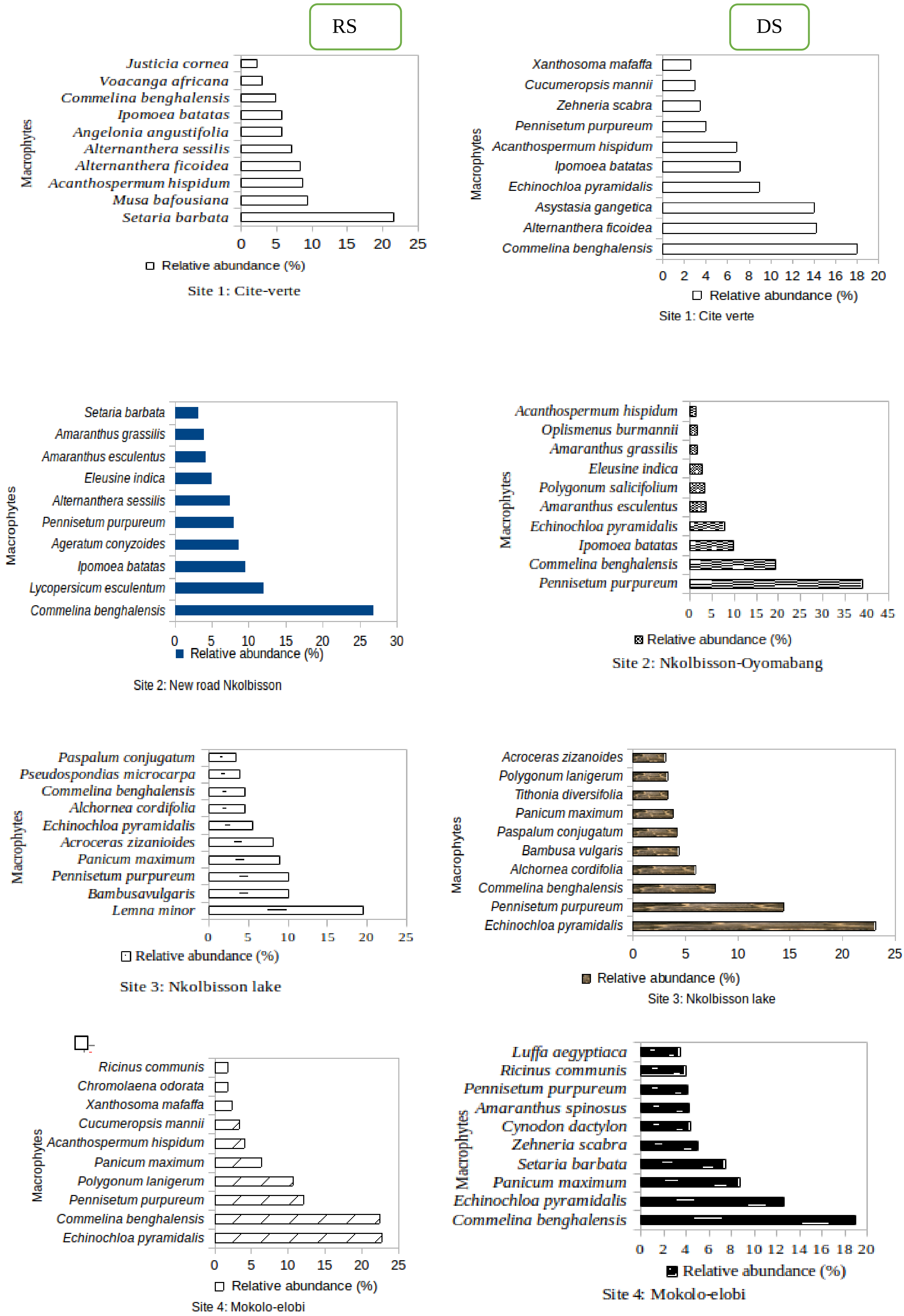

Fig. 19. Relative abundance species in each site during the rainy and dry seasons. RS: rainy season, DS: dry season (continued)

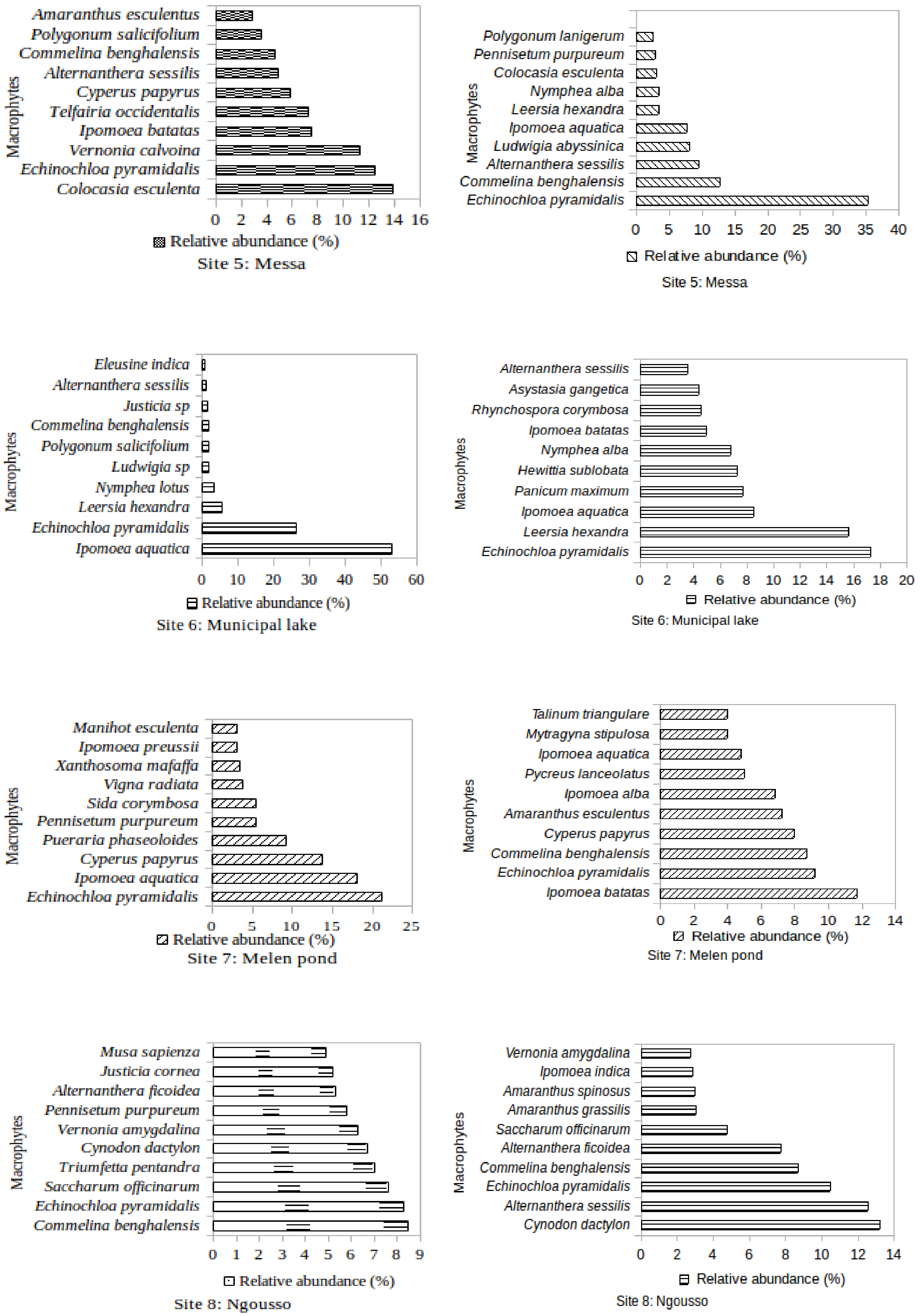

Fig. 19. Relative abundance species in each site during the rainy and dry seasons. RS: rainy season, DS: dry season (continued)

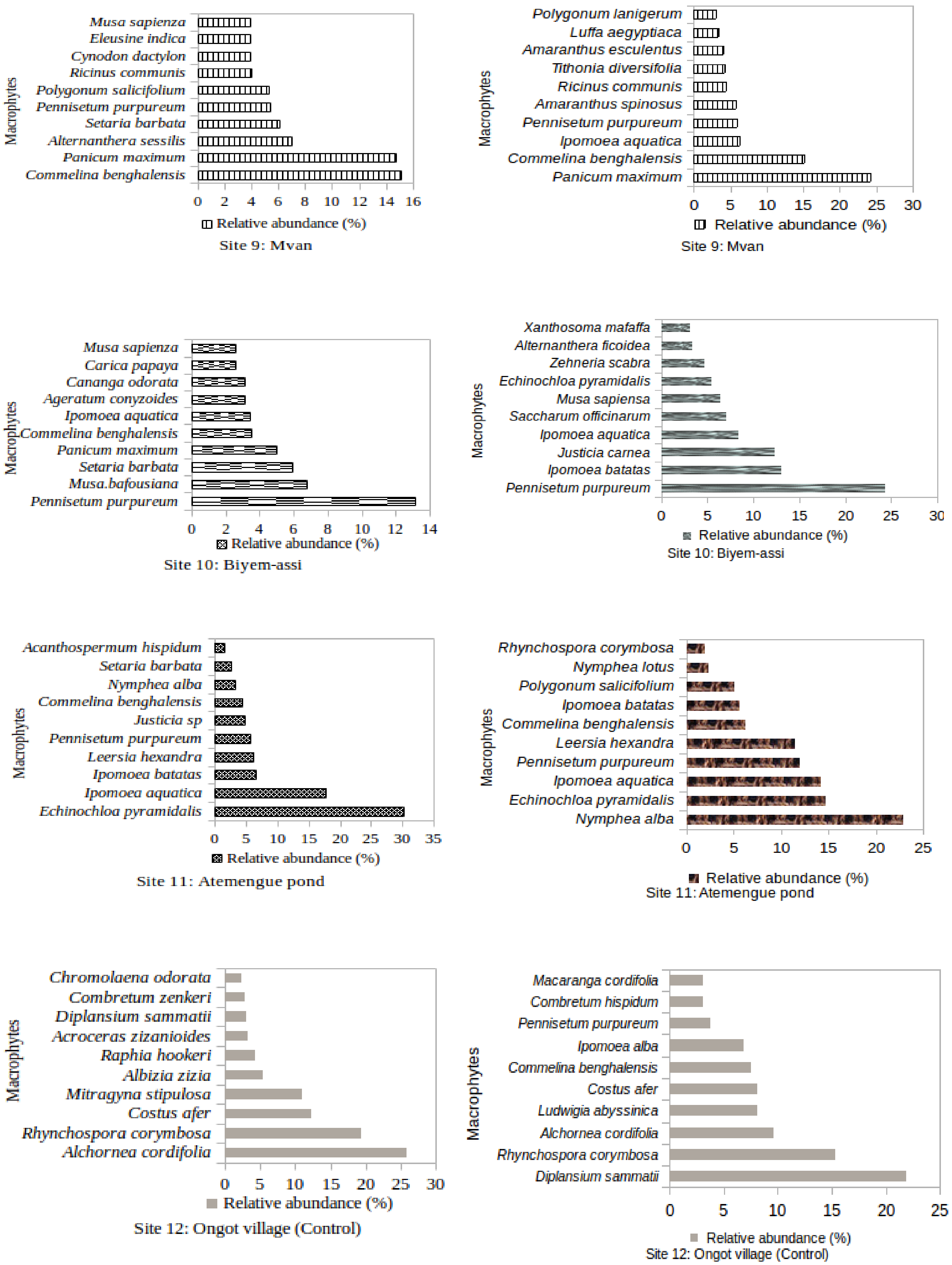

Fig. 20. Relative abundance of plants in each site during the rainy and dry seasons. RS: rainy season, DS: dry season

III.1.1.2. Distribution of pollutant-tolerant macrophytes in lowlands of Yaounde

During the rainy season, the accumulation curve produced using the rarefaction method showed a plateau. This indicates that almost all the macrophytic flora presenting bioaccumulation characteristics in the lowlands of Yaounde have been inventoried (Fig. 21a & b). The same trend was observed during the dry season specifically in each of the 12 lowlands sites (Fig. 22a & b).

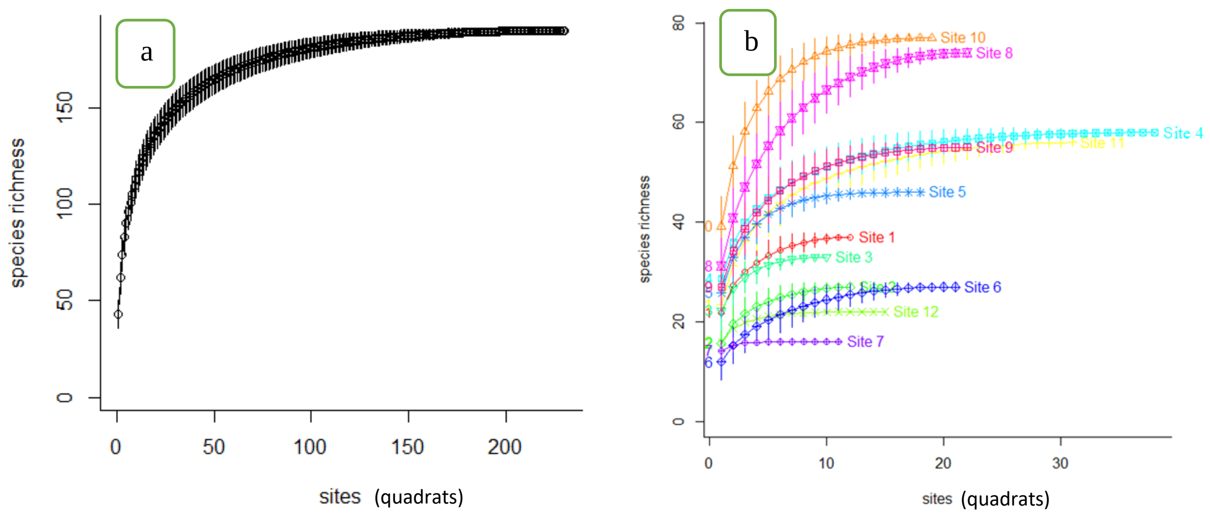

Fig. 21. Accumulation curve of the flora in lowland sites in the rainy season. (a: all sites, X axis: total number of quadrats in all sites, Y axis: total number of species identified as bioaccumulators. b: each site, X axis: number of quadrats in each site, Y axis: number of species identified as bioaccumulators in each site).

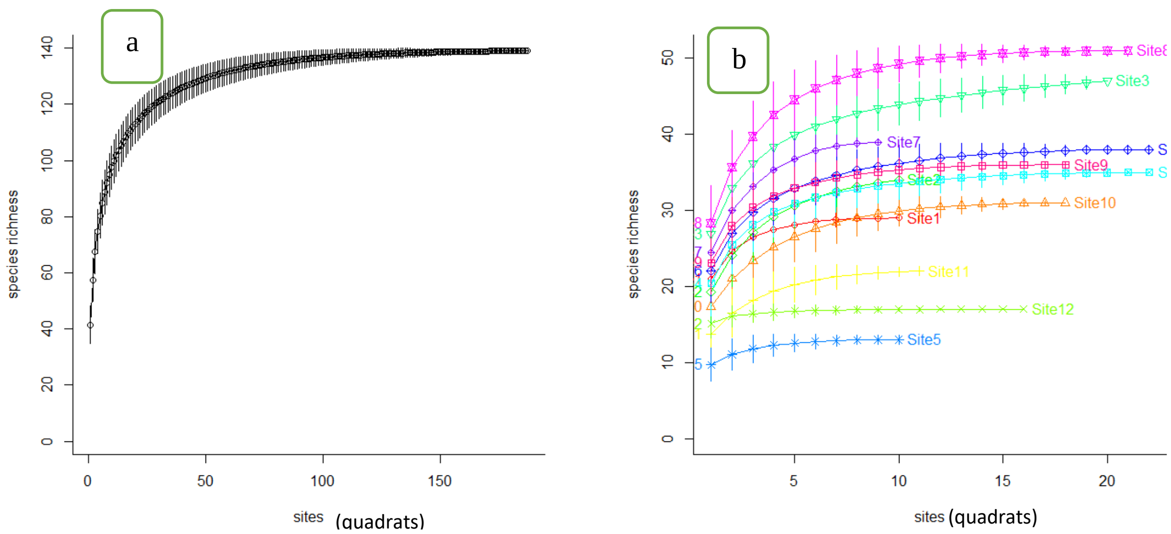

Fig. 22. Accumulation curve of the flora in lowland sites in the dry season.(a: all sites, X axis: total number of quadrats in all sites, Y axis: total number of species identified as bioaccumulators. b: each site, X axis: number of quadrats in each site, Y axis: number of species identified as bioaccumulators in each site).

The flora identified in the 12 sites during the rainy season comprised of 189 species belonging to 138 genera distributed in 63 families. On average, 21 ± 7 families were represented in the 12 sites (plus control) with a mean specie richness of 12 and a max of 36. The mean for genera was 38±15, (min. 19 and max. 70) and the mean specie richness was 44±19, (min. 22 and max. 78). During the dry season, 139 species belonging to 103 genera and dispatched in 39 families were identified. The general trends of the family richness gave an average of 17 ± 4 with a range from 12 to 23. The mean for genera and specie richness was respectively 28±8, (min. 16 and max. 40) and 32±11, (min. 13 and max. 50). Meanwhile, the number of species differed from site to site.

Depending on the seasons, the groups formed by the Principal Component Analysis (PCA) determined the floristic affinity between species in each site and similarity between the contaminated sites and the control. The Principal Component Analysis (PCA) showed three groups of species associations during the rainy season (Fig. 23):

- Group 1 (black color): the sites of this group were negatively correlated along the abscissa axis and positively along the ordinate axis, except for site 6, which was the municipal lake. The bioaccumulator species (in order of abundance) present in this site were *Ipomoea aquatica* (52.9 %), *Echinochloa pyramidalis* (26.3%), *Leersia hexandra* (5.6%), *Nymphea lotus* (3.4%) and *Ludwigia sp*. (1.9%).
- Group 2 (red color): two sites were represented in this group. They were positively correlated along the abscissa axis and site 8 (representing the lowland of Yaounde general hospital) was negatively correlated along the ordinate axis, with species such as *Commelina benghalensis* (8.5 %), *Echinochloa pyramidalis* (8.3 %), *Saccharum officinarum* (7.6 %), *Triumpheta pentandra* (7.0 %) and *Cynodon dactylon* (6.7 %).
- Group 3 (green color): sites of this group were positively correlated along the abscissa axis, except for site 11 which was the Atemengue Obili pond. Species presented in this site were *Echinochloa pyramidalis* (30.2 %), *Ipomoea aquatica* (17.6 %), *Ipomoea batatas* (6.5 %), *Leersia hexandra* (6.0 %) and *Pennisetum purpureum* (5.6 %). The three sites 6, 8 and 11, correlated negatively along the ordinate axis. The most abundant species in these sites were as follows: *I. aquatica*, *C. benghalensis* and *E. pyramidalis*.

During the dry season, the identified groups showed that:

- The first group (black color), was positively correlated along the abscissa and the ordinate axis except for site 5, which was the Messa lowland. It was completely disconnected from the other sites. The abundance of species present for this site alone were *E. pyramidalis* (35.36 %), *C. benghalensis* (12.82 %), *Althernanthera sessilis* (9.55 %), *I. aquatica* (7.82 %) and *Ludwigia abyssinica* (8.05 %).
- The second group (red color) correlated positively along the abscissa axis constituted of sites 3 and 8. Site 3 was negatively correlated along the ordinate axis and the abundant species were *E. pyramidalis* (23.11 %), *Pennisetum purpureum* (14.41 %), *C. benghalensis* (7.90 %), *Alchornea cordifolia* (5.96 %) and *Bambusia vulgaris* (4.37 %). The tendency of these abundant species growing in a similar way in these different sites may give an indication of the source and the type of pollution in this environment.

- The third group (green color) located in the center of the axis comprised of the rest of the sites. Sites 4 and 6 were far from the other sites and the abundance of species were respectively *E. pyramidalis* (22.64 %), *C. benghalensis* (22.41 %), *Pennisetum purpureum* (12.05 %), *Polygonum lanigerum* (10.59 %), *Panicum maximum* (6.41 %) and *E. pyramidalis* (17.27 %), *Leersia hexandra* (15.64 %), *I. aquatica* (8.55 %), *Panicum maximum* (7.73 %), *Hewittia sublobata* (7.27 %).

The cluster diagram also classified the sites into three groups while revealing the relation between each of them based on their diversity and recovery rate (Fig. 24).

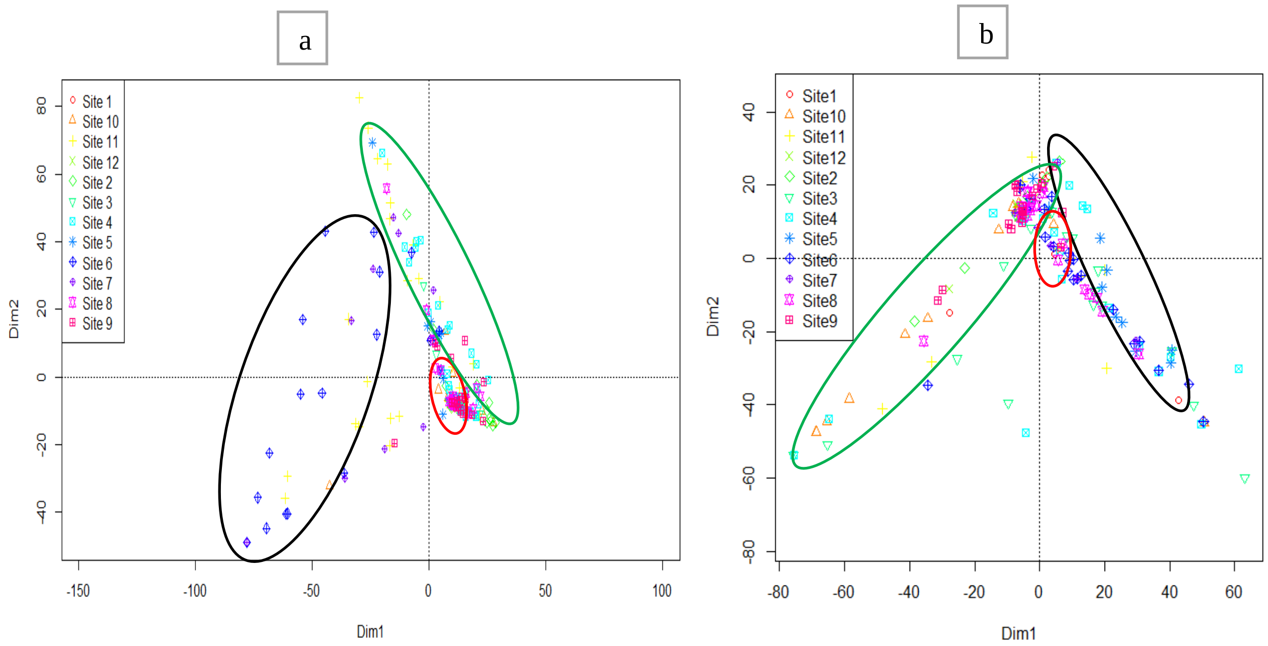

Fig. 23. Principal Component Analysis showing floristic affinities between sites. a: rainy season, b: dry season.

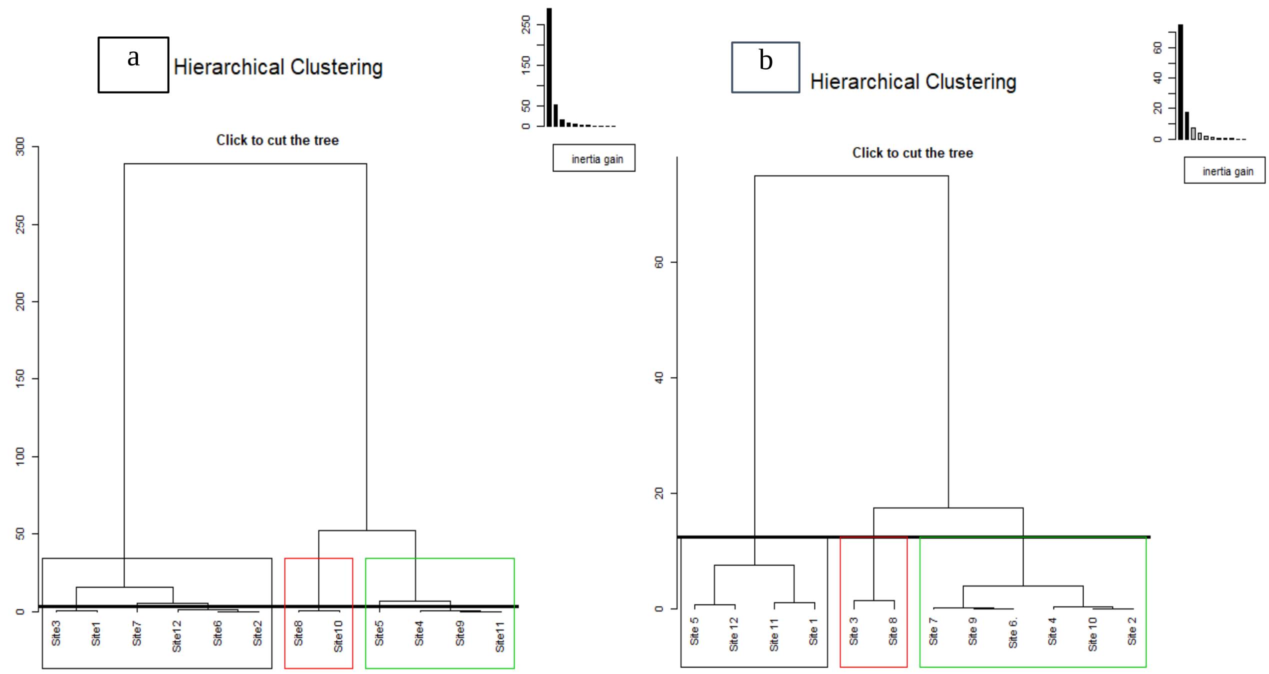

Fig. 24. Hierarchical Cluster Analysis (HCA) showing the relation between sites in groups. a: rainy season, b: dry season.

Based on their land cover and bioaccumulation characteristics in the entire floristic inventory, *E. pyramidalis*, *C. benghalensis* and *P. purpureum* were the most abundant species in the lowland areas. Furthermore, the PCA analysis and the consequent species in the groups of sites formed on the graphs for the rainy and dry seasons made it possible to classify them according to their proportions (Fig. 25).

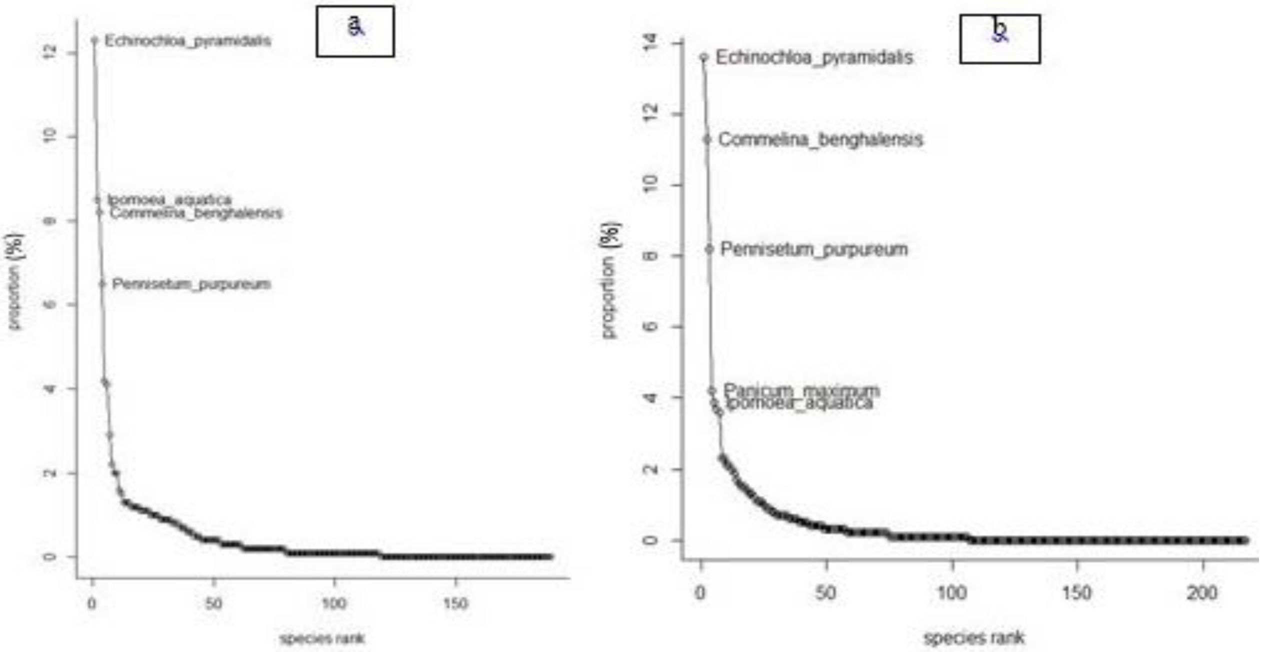

Fig. 25. Rank curve of abundant pollutant-tolerant species according to their proportion in lowland of Yaounde. a: rainy season, b: dry season.

Global analysis of PCA revealed two major components of species in the lowland sites explaining about 99.997% (rainy season) and 100% (dry season) of the cumulative variance in the data. In this study, two PCs were computed, and the variances explained by them were 99.97% and 0.027% for accumulation species collected during the rainy season; and 99.98% and 0.022% during the dry season, for all sites studied. During both seasons, species that contributed the most (PC1) were species from group 1, thereby indicating their similar characteristics and their major potential in heavy metal accumulation. They were *I. aquatica, E. pyramidalis, C.benghalensis, A. sessilis, L. abyssinica, L. hexandra, N. lotus* and *Ludwigia sp*. The second group (PC2) showed species from group 2, making them the intermediate accumulation group, with species presenting similar characteristics and bioaccumulation potential. They were *E. pyramidalis, P. purpureum, C. benghalensis, S. officinarum, T. pentandra, C. dactylon, A. cordifolia and B. vulgaris*.

With respect to seasons, there was not much variability between groups. During the rainy season, species were more numerous at the study sites than during the dry season. The distribution of the species according to their abundance and bioaccumulation characteritics in the different sites during both seasons is presented in Fig. 26.

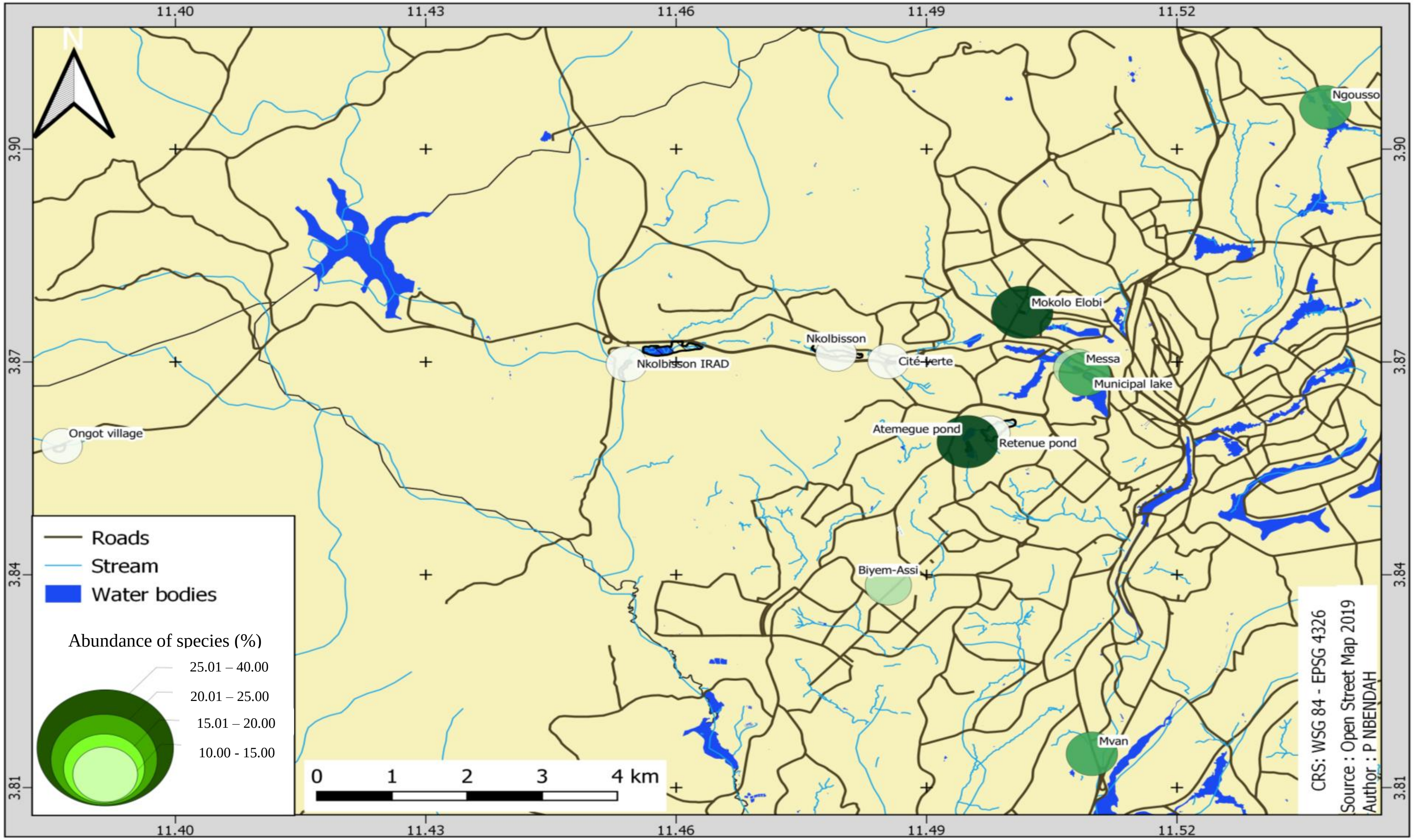

Fig. 26. Geospatial distribution of species in Yaounde lowland polluted sites and control

This map shows a sum of the species coverage identified in the quadrats (228) of each site during the dry and rainy seasons in each of the 12 lowland sites. The coverage varies according to the number of species identified as bioaccumulators present in each site and materialized on the map by circles of different diameter. Thus, the larger the diameter of the circle, the greater the number of species identified in the quadrats associated with their coverage, hence the green color observed, the intensity of which increases with the abundance of species in each site. Thus, the recovery rate of bioaccumulative species was high in 07 polluted sites out of the 11 selected lowland sites while 04 sites had a rather low recovery rate of bioaccumulative species compared to the control. The sites with high recovery of identified bioaccumulative species with a number of species in all quadrats between 25 – 40 % are site 4 (Mokolo-elobi) and site 11 (Atemengue pond). Subsequently, sites 9 (Mvan), site 6 (Municipal lake) and site 8 (Ngousso) had a number of species in the quadrats between 20 – 25 %, while sites Biyem-assi and Messa had a number between 15 – 20 %. The rest of the sites had an average coverage of identified bioaccumulative species with a number of species in the quadrats varied between 10 – 15 % compared to the control, which had a very low number of bioaccumulative species varied between 5 – 15 %.

III.1.1.5. Potentially useful macrophyte species for phytoremediation of heavy metals

Potentially useful macrophyte species that could be used in phytoremediation trials in lowlands polluted by heavy metals in Yaounde independently of season, fall into two categories. The "major" species, which were those with a relative frequency and relative abundance greater than 10% (Fri and A>10%) across quadrats of the polluted sites, and the "intermediate" species, which have a relative frequency greater than 10% (Fri >10%) and a relative abundance of between 2 and 10% (2% ≤A< 10%) in all the polluted lowlands investigated (**Erreur ! Référence non valide pour un signet.**).

Pollutant-tolerant plant species represented 24.6% of the average species recorded (164 species) during the rainy and dry seasons in the Yaounde lowlands. The family of Poaceae alone included 06 macrophyte species. Amaranthaceae and Convolvulaceae were counted two species each. Commelinaceae, Nympheaceae, Euphorbiaceae, Polygonaceae and Onagraceae were monospecific. Among species of these families, some can be grouped into two groups: grasses (such as monocotyledons) and vegetables (such as dicotyledons).

Table IIX. Pollutant-tolerant plant species of some lowland sites contaminated by heavy metals in Yaounde. (Fri: relative frequency, A: relative abundance)

Proportion (%)	Major plant species (Fri and A> 10%)	Intermediate plant species (Fri > 10%; 2%≤ A < 10%)
13.9	- *Echinochloa pyramidalis* (Lam.) Hitchc. & chase (Poaceae)	
11.6	- *Commelina benghalensis* (P. Beauv.) Kunth (Commelinaceae)	
8.5		- *Ipomoea aquatica* Forssk. (Convolvulaceae)
8.1		- *Pennisetum purpureum* (L.) (Poaceae)
4.2		- *Setaria barbata* (Lam.) Hitchc. & Chase (Poaceae).
4.2		- *Panicum maximum* Jacq. (Poaceae)
3.7		- *Ipomoea batatas* var. edulis (Convolvulaceae)
3.6		- *Alternanthera sessilis* (L.) R. Br. ex DC. (Amaranthaceae)
2.3		- *Nymphea alba* (H.) (Nympheaceae)
2.3		- *Polygonum lanigerum* R. Br. var. africanum eisn. (Polygonaceae)
2.2		- *Ludwigia abyssinica* A. Rich. (Onagraceae)
2.0		- *Alternanthera ficoidea* (L.) P. Beauv. (Amaranthaceae)
2.0		- *Leersia hexandra* Sw. (Poaceae)
2.0		- *Alchornea cordifolia* (Schumach. & Thonn.) (Euphorbiaceae)
2.0		- *Cynodon dactylon* (L.) Pers. (Poaceae)

In this study, it should be noted that during the field sampling, *I. aquatica* was not present on the selected sites during both seasons as this plant is a hydrophilic species, and for a better comparison of the potential of plants remediation in the three sites, it was replaced by *Pennisetum purpureum* the 4th species (Fig. 27).

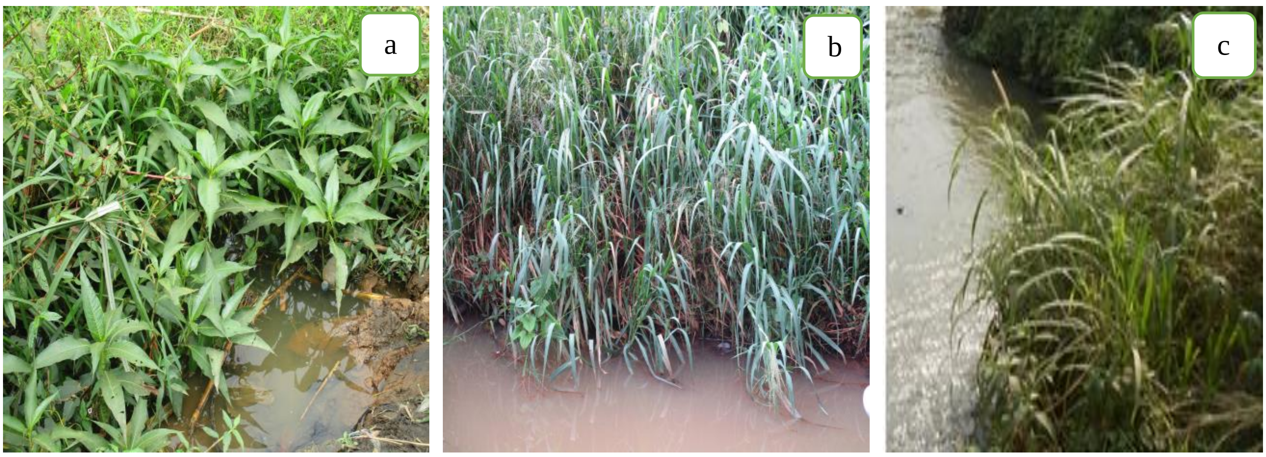

Fig. 27. Major plant species identify in the lowland areas (a. *C. benghalensis*, b. *E. pyramidalis*, c. *P. purpureum*).

Species of the group of grasses presented the highly branched root system, which are extensive, fibrous and diverse (Poaceae) compared to the group of legume (Commelinaceae) Fig. 28. Among species of these families, some have showed their potential to decontaminate heavy metal polluted soils. Table X presents some specific characteristics of the three selected plant species *E. pyramidalis* (*Ep*), *C. benghalensis* (*Cp*) and *P. purpureum* (*Pp*).

Fig. 28. Root system of the Poaceae and Commelinaceae families (a. Roots of *P. purpureum*, b. Roots system of *E. pyramidalis*, c. Roots of *C. benghalensis*).

Table X. Characteristics of selected plant species (Ngoutane et *al*., 2012; Pérez-boada et *al*., 2014; Kansagara & Pandya, 2019).

Species abreviation	Common name	Scientific name	Family and genera	Characteristics
Ep	Antelope grass	*Echinochloa pyramidalis* (Lam.) Hitchc & Chase	Poaceae, Echinochloa	Perennial herb soil attached with grow floating or submerged with creeping rhizomes that grow laterally. Erect grass species with nodes, dense, pure stand with a leaf table at 120 cm to 2 m, high potential of photosynthesis and production, growing in humid environment, good green as well as hay fodder for livestock, excellent and important source of dry season grazing forage.
Cb	Benghal day flower	*Commelina benghalensis* (L.)	Commelinaceae, Commelina	Annual or perennial herbaceous plant with ovate- lanceolate leaves. Stem of 10-30 cm tall, erect or creeping and rooting in the ground at the nodes.
Pp	Elephant grass or Napeer grass	*Pennisetum purpureum* (Schumach.)	Poaceae, Pennisetum	Robust perennial herbaceous plant with a high growth rate and biomass production. Stem of 3-4 m in height, erect large leaves 1.2 m long and 2.5 cm wide, panicle length (10-33 cm) bristly with a bottlebrush shape, high growth potential, animal fodder, solid centre of the stems, potential source of chemicals precursors and bioenergy.

III.1.2. Level of heavy metal contamination in soil, water and plant samples collected from lowland in Mokolo-elobi (site 4), Mvan (site 9) and Atemengue pond Obili (site 11) of Yaounde during the rainy and dry seasons

III.1.2.1. Soil physico-chemical and heavy metal concentrations in the three lowland sites

III.1.2.1.1. Physico-chemical properties of soils in the sites during both seasons

The physico-chemical characteristics of the soils tested in sites 4, 9 and 11 are given in Table XI. The potential hydrogen (pH) values of all the sample soils in potassium chloride (KCl) were lower than in water (H_2O) (pH (KCl) < pH (H_2O)).

Table XI. Chemical properties of lowland soils during the rainy and dry seasons. (A: soil of site 4, B: soil of site 9, C: soil of site 11).

Soil parameters		Rainy season				Dry season			
		Site 4	Site 9	Site 11	P-values	Site 4	Site 9	Site 11	P-values
		A	B	C		A	B	C	
pH (Kcl)		6.98±0.18^{b}	6.97±0.08^{b}	5.61±0.07^{a}	0.000	6.32±0.12^{b}	5.38±0.14^{a}	6.46±0.11^{b}	0.000
pH (H_2O)		7.64±0.17^{b}	7.04±0.05^{a}	6.75±0.45^{b}	0.021	6.56±0.09^{b}	6.17±0.14^{a}	6.72±0.09^{b}	0.002
EC (µS/cm)		187.97±3.65^{b}	196.7±0.1^{c}	102.2±0.26^{a}	0.000	936.67±9.61^{b}	727.33±54.6^{a}	878.67±4.04^{b}	0.001
TDS (mg/L)		187±0.0^{a}	196±0.0^{a}	102±0.0^{a}		934.67±10.02^{b}	868.67±8.08^{a}	878.33±6.80^{a}	0.000
Particle size fractions (%)	Sand (%)	77.54	73.54	47.46		81.10	73.10	47.18	
	Clay (%)	13.33	17.33	41.33		9.47	14.47	33.47	
	Silt (%)	9.14	9.14	11.21		9.42	12.42	19.35	
Organic carbon (%)		1.67	1.42	1.35		1.03	1.55	3.58	
Organic matter (%)		2.88	2.44	2.32		2.05	3.10	7.15	
CEC (meq/100g)		7.07	5.90	6.82		4.26	6.44	14.31	
Texture class		Sandy-loam	Sandy-loam	Sandy-clay		Loamy-sand	Sandy-loam	Sandy-clay-loam	

For each soil parameters of a given season values on a raw affected with different letters in upper case are significantly different at the given probability level $p < 0.05$.

The statistical analysis of the physico-chemical parameters shows a significant difference ($p<0.05$) between pH (KCl), pH (H_2O), electrical conductivity (EC) and Temperature (T) among the three sites during the rainy season. Soil samples A and B of sites 4 and 9 respectively presented the highest pH (KCl) values (6.98 and 6.97), about 1.24 units higher than soil sample C in site 11. In the same way, for pH (H_2O) soil samples A and B of sites 4 and 9 was higher respectively for 1.14 and 1 units than soil sample C in site 9. Regarding the EC value, soil sample B of site 9 and soil sample A of site 4 were respectively 1.92 times and 1.84 times elevated than EC in soil sample C of site 11.

The temperatures of soil sample B in site 9 and soil sample C in site 11 were 1.02 degrees greater than the one of soil sample A in site 4 (Fig. 29, Fig. 30). During the dry and rainy seasons, the extreme values of hydrogen potential pH (H_2O) of soil in lowland sites were slightly alkaline and acidic, (7.04 - 7.64 units) and (6.17 - 6.72 units) respectively. A different trend was observed in the rainy and dry seasons for pH (KCl), where the pH values were acidic, respectively (5.61 – 6.98 units) and (5.38 – 6.16 units). Statistical analysis showed a significant difference ($p<0.05$) for pH (KCl), pH (H_2O), electrical conductivity (EC) and total dissolved solid (TDS) alongside the three sites during the dry season.

Compared to the dry season, pH (KCl) in soils A and C was 1.18 and 1.2 units respectively higher than their value in soil B. The same trend was observed for pH (H_2O) where soils A and B were 1.1 units higher than the one in soil B. EC and TDS in soils C and A were 1.2 and 1.3 times and 1.1 % and 8 % respectively elevated than EC and TDS in soil B. Among soil properties, soil pH has strong effects on solubility and speciation of metals both in the soil and particularly in the soil solution (Fanrong et *al*., 2011). During the rainy season, soils were slightly alkaline in all the three sites. However, in the dry season, soils become acid due to the impacts of anthropogenic activities on these sites.

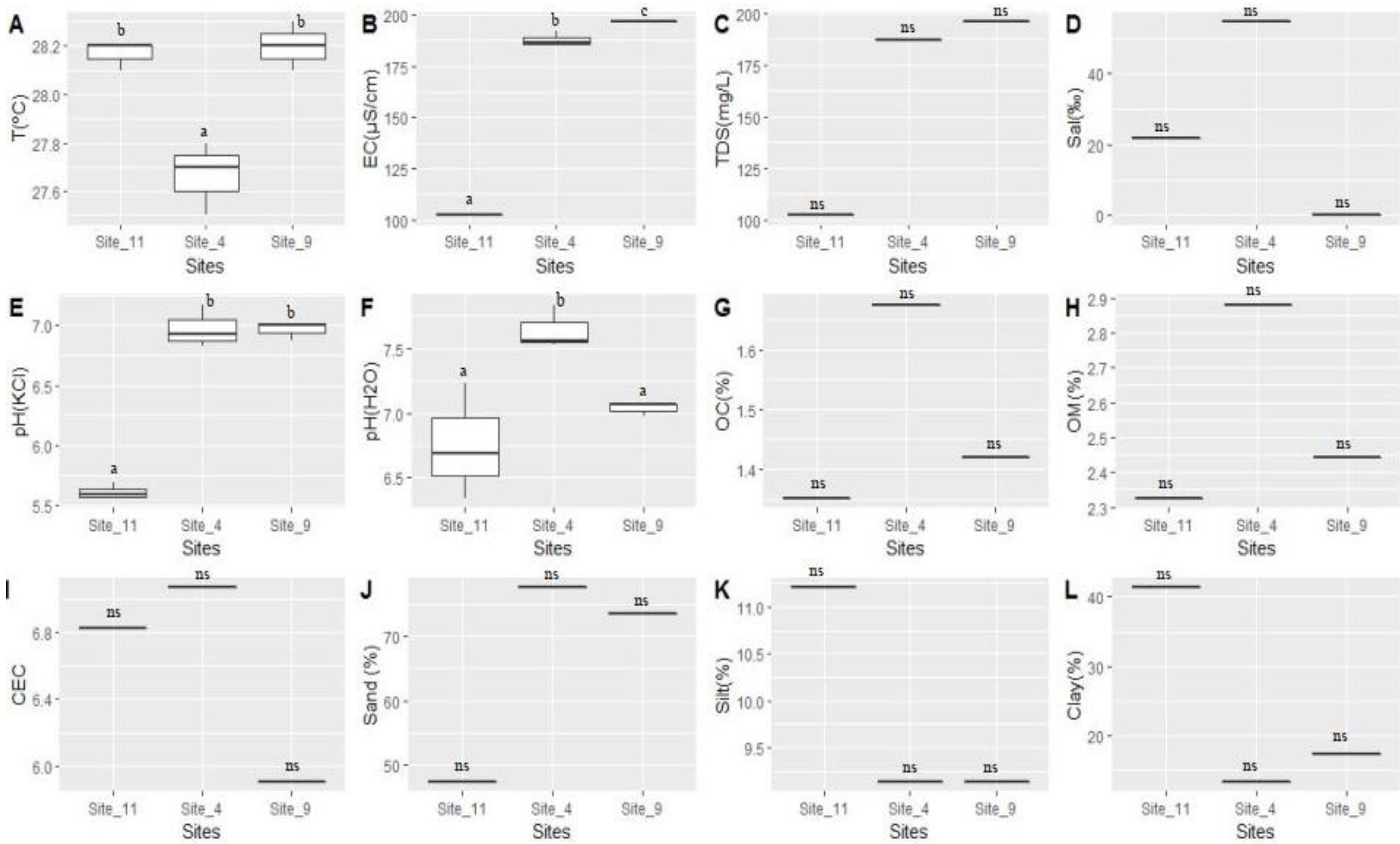

Fig. 29. Physico-chemical properties of soils in sites 4, 9 and 11 during the rainy season (RS). ([ns] and [a], [b], [c] statistically not significant and significant at $p<0.05$).

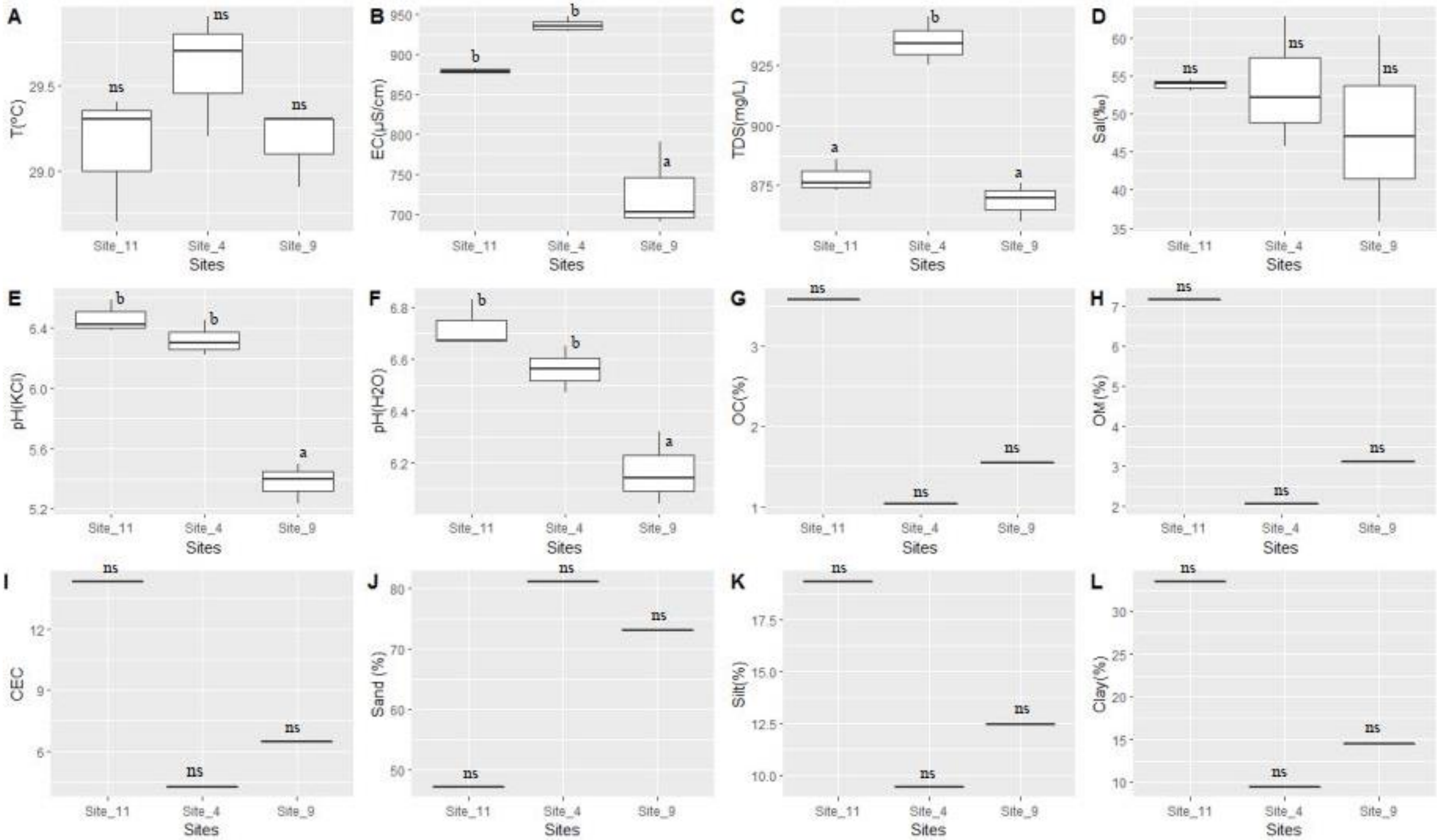

Fig. 30. Physico-chemicals parameters of soils in different sites during the dry season (DS). ([ns] and [a and b] statistically not significant and significant at $p<0.05$).

According to the soil texture classification system in the United States Department of Agriculture (USDA) (Groenendyk et *al*., 2015), the textural analysis during the rainy season showed that soil samples A and B of sites 4 and 9 were sandy-loam compared to soil sample C of site 11 which was sandy-clay. Contrastingly, in the dry season, soil textural characteristics ranged respectively from loamy sand, sandy loam and to sandy clay loam for sites 4, 9 and 11. Grain-size distribution of soils showed several differences between the investigated sites of the Mfoundi watershed. In all the soil samples during the rainy and dry seasons, sand fraction prevailed with the maximum value in soil A of site 4 (63.38% and 71.89%) higher than sand in soil C of site 11. Therefore, the clay fraction in soil A of site 4 was 3.15 times higher than clay in soil A of site 4 during the rainy season compared to the dry season where it was found 3.5 times higher in soil C of site 11 than in soil A of site 4. The contribution of silt in the rainy season in soil C of site 11 was 22.87% higher than soils A and B of sites 4 and 9 during dry season for soil C, site 11 (Fig. 31).

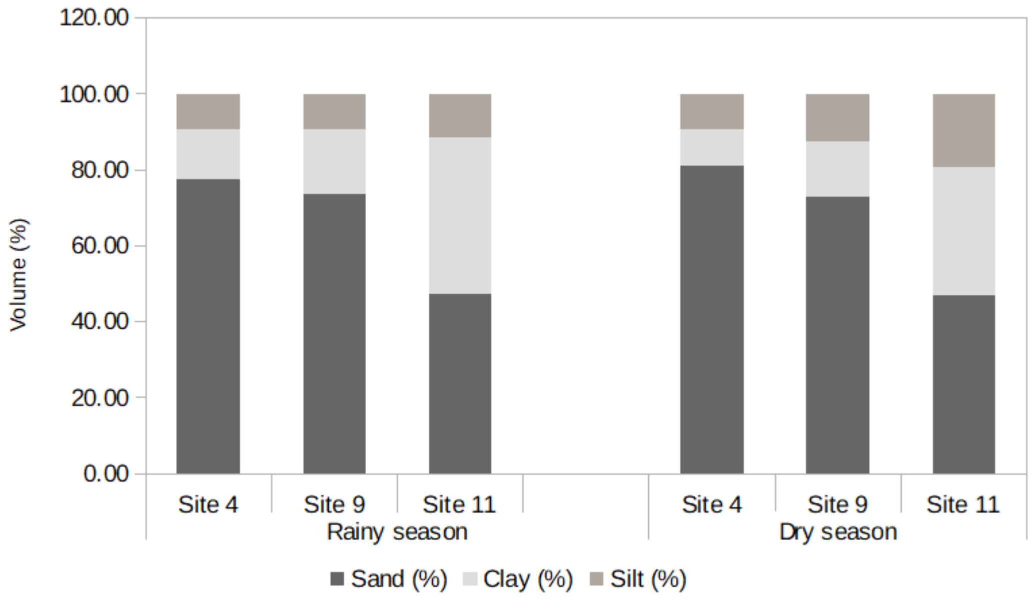

Fig. 31. An average percentage of different fractions (clay, silt and sand) in the soils of the three investigated sites of the lowlands during the rainy and dry seasons

Just as it is for aquatic ecosystems, the upper part of soils (sediments) acted as major sinks for metals, further assessment of the natural environment of the region included granulometric description, total organic carbon (TOC) in soils and metal contents. In this study, the level of total organic carbon (TOC) in soils during the rainy season was 1.48 ± 0.14% and 2.05 ± 1.16%. During the dry season, the corresponding TOM values were 2.55 ± 0.24% and 4.11 ± 2.34% in both seasons.

III.1.2.1.2. Heavy metal concentrations and norms applied to soils

The results of the heavy metal concentrations (Pb, Cd, Cr, Ni, Zn, Cu, As, and Co) in soil samples from different sites during the rainy and dry seasons are summarised in Table XII. In all sampled soils, the mean concentrations of Pb, Cd, Zn, Cu and As were below the WHO threshold values, while the mean concentrations of Cr in all sites exceeded these values. These values were 117 (µg/g), 203.5 (µg/g) and 407.25 (µg/g)>100 or 150 µg/g respectively (Anonymous 6, 2007; Adesuyi et *al*., 2018). Concerning Ni and Co, the highest concentration was observed only in site 11(C) in addition to site 4(A) for Co and the values were higher than the norm values (80.29 µg/g) > 50 µg/g and (8.17; 20.23) µg/g > 8 µg/g respectively. The amounts of Cd and As were the lowest compared to the other metals, independent of the soil types. These low values of Cd and As in the soil could be attributed to the low industrial activities in the area. Arsenic is often strongly bound to iron, aluminium and calcium compounds, and remains relatively more mobile in sandy soils poor in organic matter. However, the high concentration of Cr could be due to activities such as garages with old batteries, fertilisers, hospital effluents and industrial effluents around lowland sites.

Table XII. Heavy metal concentration in soils and norms applicable to soils. (* (Anonymous 6, 2007); ** (Anonymous 11, 2002), A: soil of site 4, B: soil of site 9 and C: soil of site 11).

Heavy metals (µg/g)	Site 4 A	Site 9 B	Site 11 C	P-values	Norms applied (Anonymous 6, 2007) (µg/g)	(Anonymous 11, 2002)
Pb	38.89 ± 4.94^{b}	59.08 ± 13.55^{b}	8.76 ± 1.57^{a}	0.003	100.00 *	100 **
Cd	0.005 ± 0.002^{a}	0.005 ± 0.002^{a}	0.01 ± 0.004^{a}	0.984	3.00 *	3.00 *
Cr	117.52 ± 13.93^{a}	203.52 ± 2.17^{ab}	285.13 ± 62.70^{b}	0.017	100 *	150 **
Ni	13.13 ± 2.13^{a}	18.89 ± 1.87^{a}	80.29 ± 24.88^{b}	0.009	50.00 *	75.00 **
Zn	125.12 ± 13.05^{a}	112.17 ± 1.02^{a}	156.23 ± 36.13^{a}	0.377	300 *	300 **
Cu	75.94 ± 10.71^{b}	47.78 ± 1.53^{a}	28.72 ± 3.29^{a}	0.000	100 *	140 **
As	4.73 ± 2.11^{a}	7.43 ± 3.32^{a}	09.57 ± 4.28^{a}	0.987	20 *	-
Co	8.17 ± 0.619^{a}	7.65 ± 0.016^{a}	20.23 ± 1.71^{b}	0.000	8	-

For each soil parameters of a given season values on a raw affected with different letters in upper case are significantly different at the given probability level $p < 0.05$).

III.1.2.1.3. Effects of sites and seasons on the distribution of heavy metal in soils

The heavy metal analysed showed a significant difference ($p<0.05$) between the three sites studied for Pb, Cu, Cr, Ni and Co. The results showed that, Cr and Ni in sites 9(B) and 11(C) were 1.7, 2.6 and 1.44, 6.11 higher than Cr and Ni in site 4(A) respectively. However, the content of Co in site 4(A) and 11(C) was around 7% and 165% higher than Co in site 9(B). Furthermore, Cu was observed to be higher in sites 4(A) and 9(B) than Cu in site 11(C) respectively by 164% and 66%. For Pb, sites 9(B) and 4(A) presented 7- and 4-times higher concentrations than site 11. This showed that metal contents varied according to the topography of the environment and the types of human activities around the sites.

For each season, the statistical analysis showed no significant difference ($p>0.05$) between the most heavy metals analysed (Pb, Cu, Cr, Zn, Co and Ni). However, for As and Cd, a significant difference ($p<0.05$) was found, even though they were all below the threshold values.

III.1.2.1.4. Correlation between physico-chemical properties and heavy metals in soils

Across the three soils (A, B and C), highly significant positive correlations ($p<0.01$) were observed between pH(KCl) and pH(H_2O) with (r = 0.732**), pH(H_2O) with TDS, T, Cu, As respectively with r = -0.642**, -0.735**, 0.591**, -0.609**, EC and TDS, Sal, T, As, Cd (r = 0.989**, 0.617**, 0.895**, 0.891**, 0.895), TDS and Sal, T, As, Cd (r = 0.616**, 0.901**, 0.911**, 0.881**), T and As, Cd (r = 0.773**, 0.756**), CEC and TOC, TOM, Sand, Silt, Cr, As, Ni, Cd, Co, Zn (r = 0.991**, 0.977**, -0.708, 0.929**, 0.946**, 0.975**, 0.662**, 0.904**, 0.899**). The same trend was observed for TOM, physico-chimical parameters and metals, while sand was correlated with all metals except As, Cd and Zn. Clay correlated with Pb, Cu and Co (r = -0.643**, -0.678** and 0.828**). Silt was strongly correlated with all metals except Pb and Cu. Cr was strongly correlated with As, Ni, Cd, Co, Zn ($r \geq 0.681$**), while As was with Cd and Ni (r = 0.615** and 0.968**), Ni was with Cd, Co and Zn (r = 0.732**, 0.894** and 0.882), while Cd and Co were correlated with Zn (r = 0.655** and 0.664**) respectively. Another significant correlation ($p<0.05$) was also found with physico-chimical parameters and metals (Fig. 32). The significant positive correlations between metals, suggest similar sources (geogenic or anthropogenic) of metal inputs in soils.

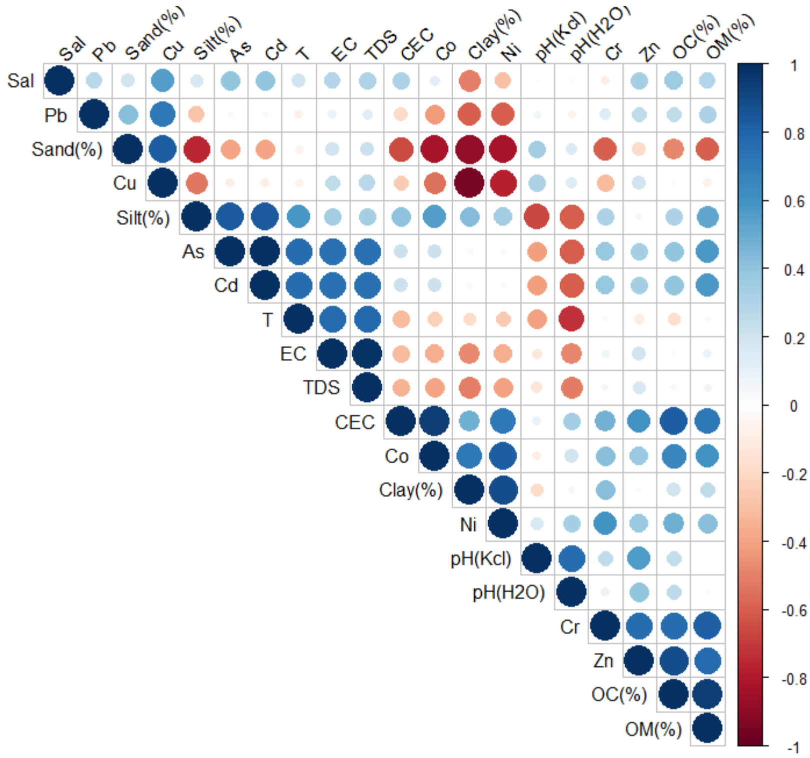

Fig. 32. Correlation coefficient among heavy metals and physico-chemical characteristics of soils. Significant Pearson correlations ($p < 0.01$) between heavy metals and physico-chemical properties of soils. Positive correlations were displayed in blue and negative correlations in red see (Appendix 6) for all correlation values and their significance. The correlogram represents the correlations for all pairs of variables. The intensity of the color was proportional to the correlation coefficient so the stronger the correlation (i.e., the closer to -1 or 1). The color legend on the right hand side of the correlogram shows the correlation coefficients and the corresponding colors.

III.1.2.1.5. Assessment of toxic metal pollution in lowlands soils

III.1.2.1.5.1. Contamination levels of heavy metals

In this study, Cr appeared as the most heavily contaminated element in all the soils. Table XIII presented the Igeo values for each metal in soil. For lead (Pb), cadmium (Cd), nickel (Ni), Zinc (Zn), arsenic (As) and cobalt (Co), the geo-accumulation index values were less than zero and less than 1 (0>Igeo<1) regardless of all soil types. Therefore, the soils investigated were qualified as uncontaminated by Cd, Ni, Zn, As, Co and Pb. For copper (Cu), the geo-accumulation index was higher than 0 and less than 1 (0<Igeo<1). The soils were qualified as moderately contaminated by Cu with an Igeo value of 0.473. Chromíum (Cr) showed the highest Igeo value for the soils. The results showed that the Igeo value of Cr was higher than 2 and less than 3 (2<Igeo<3). The soils were moderately qualified as heavily contaminated with an Igeo value of 2.308.

In decreasing order, the contamination of the soils was ranged as follows: Cr>Cu>Co>Zn>Pb>Ni>As>Cd. Consequently, it can be observed that soil C and B were respectively more contaminated with Cr and Cu in this study.

Table XIII. Geo-accumulation index (Igeo) of heavy metals in lowland soils. (A: soil of site 4, B: soil of site 9, C: soil of site 11).

Heavy metals (µg/g)	A	B	C	Mean	Contamination level	Control back-ground (Bn)
Pb	0.065	0.668	-2.084	-0.450	-2.08<Igeo<0.67 Uncontaminated	24.78
Cd	-9.76	-8.912	-8.633	-9.103	-9.76<Igeo<-8.63 Uncontaminated	2.36
Cr	1.614	2.412	2.898	2.308	1.61<Igeo<2.89 Heavily contaminated	25.5
Ni	-2.088	-1.563	0.525	-1.042	-2.09<Igeo<0.52 Uncontaminated	37.2
Zn	-0.262	-0.419	0.059	-0.207	-0.42<Igeo<0.06 Uncontaminated	100
Cu	1.163	0.495	-0.239	0.473	-0.24<Igeo<1.16 Moderately contaminated	22.6
As	-1.829	-1.176	-0.811	-1.272	-1.83<Igeo<-0.81 Uncontaminated	11.2
Co	-0.443	-0.540	0.864	-0.040	-0.54<Igeo<0.864 Uncontaminated	7.41

III.1.2.1.5.2. Pollution distribution and sources of heavy metals concepts

The pollution levels of the lowland soils were assessed using the pollution index (PI). Fig. 33 shows the variation of the PI of metals in the soils of different sites compared to the permissible threshold values. As indicated in the figure, all soils were polluted with Cr, Zn, Co, Pb and Cu (PI>1). Cd and As pollution in the soils were not significant (PI<1). Therefore, regardless the soil sites, the PI values of lead (Pb) were higher than 1 (PI>1) in soils A and B indicating the Pb-pollution of these soils. However, in soil C, the PI was less than 1 (PI<1) indicating no Pb-pollution. On the contrary, the PI value of nickel (Ni) was higher than 1 (PI>1) in soil C and lower than 1 (PI<1) in soils A and B. This presented Ni-pollution in soil C. The PI values follow the decreasing order: Cr>Cu>Co>Pb>Ni>Zn>As>Cd.

The assessment of the pollution level of each soil sample is represented in Table XIV by the IPI values. The ranges of the integrated pollution index (IPI) values were generally higher than 3 (IPI≥3) in all soil sites, indicating the higher pollution of soils. The different types of lowland soils can be classified by decreasing IPI values as follows: C (8.06)>B (5.79)>A (3.41).

Consequently, all lowland soils were categorized as heavily polluted (IPI>3). Meanwhile, the potential ecological risk index (RI) was adopted to assess the ecological risk levels of metals in the current soils.

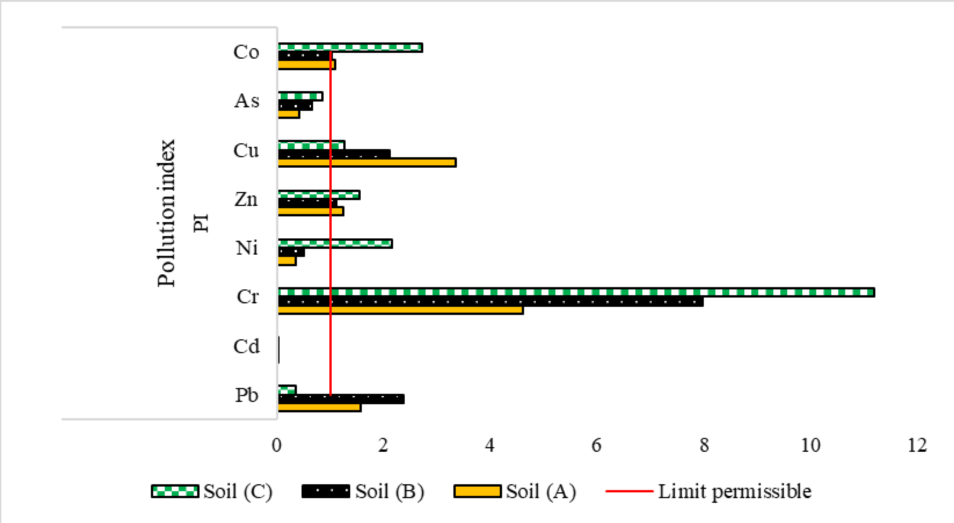

Fig. 33. Variation of the pollution index (PI) of heavy metals in the three lowlands soils. (Soil A: soil of site 4, soil B: soil of site 9 and soil C: soil of site 11).

III.1.2.1.5.3. Potential ecological risk assessment

The potential ecological risk index (RI) of eight heavy metals (i.e., Pb, Cd, Cr, Ni, Zn, Cu, As and Co) in lowlands has been widely used in ecological risk assessment of the pollution soils as shown in Fig. 34. According to grading standards, the single potential ecological risks index (E^i_r) of potentially toxic metals in surface soils are listed in the following decreasing order of Cr>Cu>Co>Pb>Ni>As>Zn>Cd.

Table XIV. Distribution of pollution index (PI) and integrated pollution index (IPI) in lowland soils. (A: soil of site 4, B: soil of site 9, C: soil of site 11).

Index	Heavy metals (µg/g)	A	B	C	Control background (Bn) Cameroon (Defo et *al.*, 2015) (µg/g)	Control background (Bn) China (Binggan & Linsheng, 2010) (µg/g)
PI	Pb	1.57	2.38	0.35	24.78	26
	Cd	0.002	0.003	0.004	2.36	0.097
	Cr	4.60	7.98	11.18	25.5	61
	Ni	0.35	0.51	2.16	37.2	26.9
	Zn	1.25	1.12	1.56	-	100
	Cu	3.36	2.11	1.27	-	22.6
	As	0.42	0.66	0.85	-	11.2
	Co	1.103	1.032	2.73	-	7.41
IPI		3.41	5.79	8.06		

The single potential ecological risks index (E^i_r) values of the metals were all lower than 40, indicating that there might be a slight risks from these metals in lowland soils. The total potential ecological risks (RI) were in decreasing order from site 11 to site 4 (11>site 9>site 4). The RI values were 46.64, 54 and 65.15 respectively for sites 4, 9 and 11 lower than 150, demonstrating that there was a slight level of risk in the lowland soils studied. Thus, Cr with the E^i_r (22.36) was the metal, which posed a potentially considerable risk to the lowland soils studied, followed by Cu with 16.8 E^i_r.

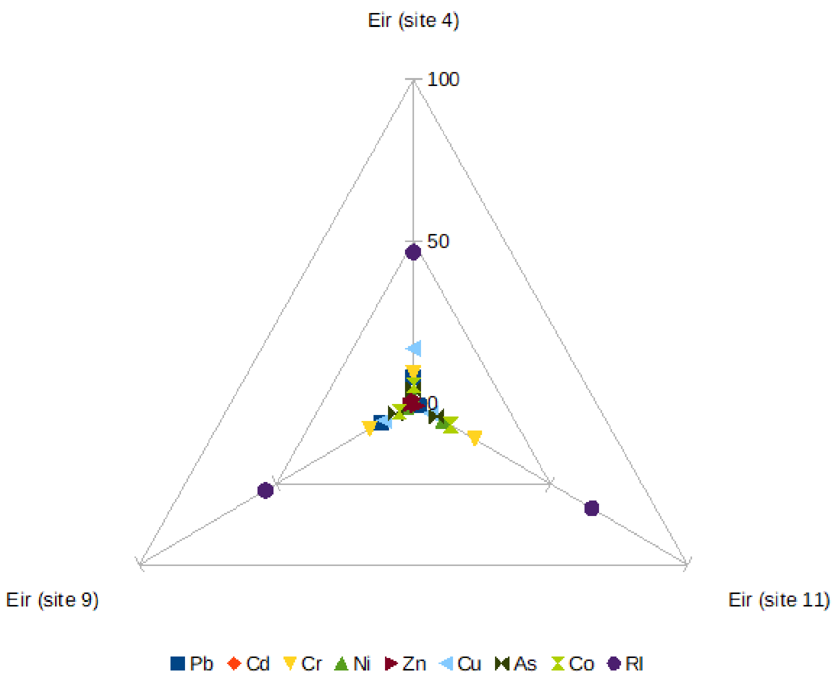

Fig. 34. Ecological risk of potentially toxic metals in lowland soils sites studied. (E^i_r): single potential ecological risk index, RI: potential ecological risk index).

III.1.2.1.6. Source analysis of heavy metals in lowland soils

In the present study, two essential components were discovered while executing the PCA survey, and extracted with respect to Eigen values greater than 1 (Fig. 35). The PCA results corresponded well with the obtained correlation coefficients. From the present examination, the initial two main parts were figured in PCA and the fluctuation clarified by them was 57.97% and 23.35% of total variance. PC1 and PC2 together explained 81.32 % of the total variance, indicated that the lithogenic factor (parent material) dominated the distribution of most parts of the considered metals in the study. The high level of Ni, Cr, Co, Zn, As and Cd were processed, characterizing the main part (PC1) accounted for 57.97%. The strong positive loading exceeded 0.7 between the elements of this group and they showed strong positive relationship. The second vital segment (PC2) accounted for 23.35% of the total variance incorporated Cu and Pb suggested similar anthropogenic source. Therefore, Pb presented a strong positive loading (0.91) compared to Cu (0.53), and both elements showed moderate positive correlation (r=0.492*), suggesting that the sources of Pb in the lowland soil sites could be both geogenic and anthropogenic.

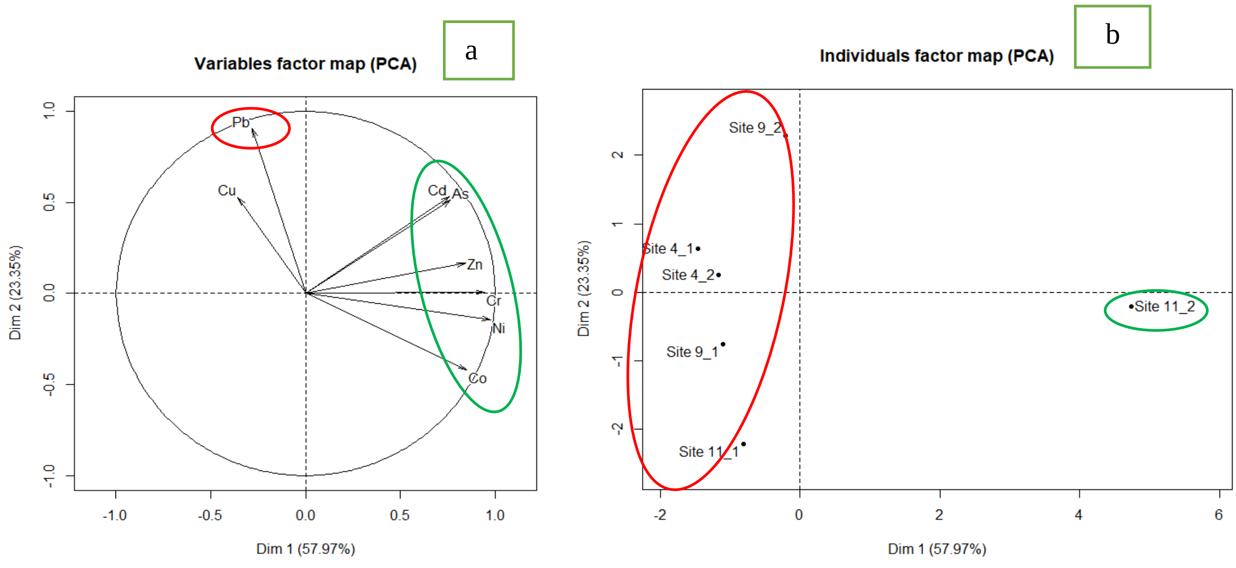

Fig. 35. Principal component analysis (PCA) of heavy metals in soils of sites 4, 9 and 11 (_1: rainy season, _2: dry season).

III.1.2.1.7. Correlation between pollution indices and heavy metals in lowland soils

The eco-toxicity of metals in lowland soils was assessed according to the contents of each metal in the three soils of the study sites. A correlation analysis between the geo-accumulation index (Igeo), the integrated pollution index (IPI) and the potential ecological risk index (RI) results was carried out. The results did not show any significant relationship between all factors, thereby explaining the low overall level of soil contamination obtained. On the other hand, the IPI presented a significant positive relationship with Cd, Cr and As with coefficients of 1.000** and 0.999* respectively (Table XV). These factors predicted the distribution, bioavailable concentration and speciation of these metals in soils. However, tropical lowland soils with weathering processes affected the vulnerability of soils to pollution. According to Yerima et *al*. (2014), the weathering processes generate acid soils with low buffering capacity and promote the retention of heavy metals in their exchange complex. The pollution of soil with Cu is an indication that soil is also polluted with other heavy metals.

This case study, in which several quantitative risk assessment index analysis were applied (potential ecological risk factors, geoaccumulation index, integrated pollution index), revealed no ecological impact of the studied metals (Pb, Cd, Cr, Ni, Zn, Cu, As and Co) on lowland ecosystems. Furthermore, the contamination information obtained when using E^i_r and other methods could be a warning about the prevailing situation in the lowlands and future solutions for the management of this ecosystem.

Table XV. Correlation between pollution indices of soils and heavy metals in the lowland soils. (IPI: Integrated pollution index, Igeo: geoaccumulation index and RI: potential ecological risk index)

	Pb	Cd	Cr	Ni	Zn	Cu	As	Co	IPI	Igeo	RI
Pb	1										
Cd	0.540	1									
Cr	0.473	0.695*	1								
Ni	0.172	0.554	0.918**	1							
Zn	0.400	0.969**	0.738*	0.654	1						
Cu	0.838**	0.550	0.475	0.333	0.479	1					
As	0.647	0.668*	0.943**	0.860**	0.672*	0.691*	1				
Co	0.337	0.556	0.924**	0.970**	0.632	0.525	0.927**	1			
IPI	-0.586	1.000**	1.000**	0.897	0.676	-0.995	0.999*	0.840	1		
Igeo	0.282	0.601	0.614	0.200	-0.170	-0.687	0.653	0.084	0.612	1	
RI	-0.687	0.993	0.991	0.947	0.766	-0.974	0.983	0.904	0.991	0.503	1

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

III.1.2.2. Water physico-chemical and heavy metal concentrations in the lowland sites

III.1.2.2.1. Physico-chemical characteristics of water during the rainy and dry seasons

The parameters of water quality recorded in all the three sites studied are presented in Table XVI. All the physico-chemical parameters varied significantly ($p<0.05$) from one site to another. During the rainy season, the temperature (T) of lowland waters varied from 24.8 to 28.3ºC, and increased slightly from 25.07 to 28.83 ºC during the dry season. Extreme potential of hydrogen (pH) values of waters in the lowland sites were slightly alkaline (7.58 - 8.26) units in the rainy season and (7.37 - 7.93) units in the dry season. With regard to water quality, the minimum pH value in the study sites was higher than the interval required by MINEPDED and WHO respectively (6 – 9) and (6.5 – 8.4) for wastewater used for irrigation (Anonymous, 2008, Anonymous, 2017). During the rainy season, electrical conductivity (EC) ranged from 216 to 502.3 $\mu S.cm^{-1}$, and total dissolved solids (TDS) from 212 to 504.67 $mg. L^{-1}$ compared to the dry season, which were respectively from 316 to 629.3 $\mu S.cm^{-1}$ and 297.3 to 626.3 $mg. L^{-1}$.

Table XVI. Physico-chemical characteristics of the lowland waters in sites 4, 9 and 11 and norms applied. (* WHO (Anonymous, 2017); ** WHO (Anonymous, 2003); ***MINEPDED (Anonymous, 2008)).

Water parame-ters	Rainy season				Dry season				Norms
	Site 4	Site 9	Site 11	P-value	Site 4	Site 9	Site 11	P-value	
T (ºC)	26.97 ±0.06^{ab}	28.3 ±0.72^{b}	24.8 ±1.11^{a}	0.004	26.97 ±0.15^{a}	28.83 ±1.5^{a}	25.07 ±2.00^{a}	0.052	30***
EC (µS/cm)	345 ±1^{ab}	502.3 ±3.78^{b}	216 ±14^{a}	0.000	452.3 ±18.5^{ab}	629.3 ±31.5^{b}	316 ±44.7^{a}	0.000	3000**
TDS (mg/L)	345.3 ±1.5^{ab}	504.67 ±0.58^{b}	212 ±14.8^{a}	0.000	450.3 ±14.57^{ab}	626.3 ±23.7^{b}	297.3 ±57.27^{a}	0.000	30*
pH	7.58 ±0.00^{a}	8.26 ±0.5^{b}	8.0 ±0.15^{ab}	0.074	7.60 ±0.03^{a}	7.93 ±0.37^{a}	7.37 ±0.28^{a}	0.109	6.5 -8.4* 6 - 9***
Sal (‰)	0.2 ±0.00^{a}	0.1 ±0.00^{a}	0.00 ±0.00^{a}		0.2 ±0.00^{b}	0.1 ±0.00^{ab}	0.067 ±0.06^{a}	0.007	-
Eh (mV)	-18.2 ±1.6^{ab}	-72.2 ±24.35^{a}	11.87 ±5.8^{b}	0.001	-20.4 ±2.8^{ab}	-141.67 ±68.4^{a}	3.43 ±9.25^{b}	0.009	-

For each water parameters of a given season values on a raw affected with different letters in upper case are significantly different at the given probability level $p < 0.05$.

III.1.2.2.1.1. Effects of sites on the variation of water physico-chemical parameters

The interaction between season and the sites showed that, there was a significant difference between site and season (sites×seasons= 0.047<0.05), and between the different sites (site4 × site9 × site11= 0.0001 < 0.05). The statistical analysis with one-ways ANOVA also showed the significant difference between sites 4, 9 and 11 for all the physico-chimical parameters. The results showed that for each parameter analysed (pH, T, EC, TDS, Sal and Eh), P was lower than 5% (P<0.05) (Table XVII). The pH values of sites 11 and 9 were respectively 1 and 1.1 units higher than the pH of site 4. Temperature of water at sites 4 and 9 was respectively 2.03 and 3.03 degrees higher than in site 11. The same trends were observed for EC and TDS, where their values in site 4 (43% and 52%) and 9 (99% and 111%) were respectively higher compared to site 11. Conversely, the potential redox (Eh) in site 11 was 8 times higher than in sites 4 and 9, where Eh was practically zero. Salinity (Sal) was 200% and 500% higher in sites 9 and 4 respectively than in site 11.

III.1.2.2.1.2. Effects of season on the variation of the physico-chemical parameters of water.

The comparison of seasons using ANOVA showed that, between the rainy and dry seasons, there was no significant difference between the analysis of the physico-chemical parameters (T, EC, TDS, Eh, Sal and pH) where p (0.069) was higher than 5 % ($p>0.05$) (Table XVII). In this study, the influence of the seasons on the physico-chimical parameters was negligible.

Table XVII: Season and site effects on the physico-chemical parameters of water in lowland sites. (RS: rainy season, DS: dry season).

Water parameters	Seasons effects		Sites effects			
	RS	DS	Site 4	Site 9	Site 11	P-values
T (ºC)	26.7 ±1.67^{a}	26.95 ±2.06^{a}	26.97 ±0.1^{a}	28.56 ±1.1^{b}	24.93 ±1.46^{c}	0.000
EC (µS/cm)	354.4 ±124.4^{a}	465.9 ±139^{a}	398.67 ±59.94^{ab}	565.83 ±72.4^{b}	266 ±62.27^{a}	0.000
TDS (mg/L)	354 ±127.1^{a}	458 ±146^{a}	397.83 ±58.25^{ab}	565.5 ±68.3^{b}	254.67 ±59.87^{a}	0.000
pH	7.95 ±0.39^{a}	7.63 ±0.34^{a}	7.59 ±0.19^{a}	8.1 ±0.44^{b}	7.69 ±0.39^{ab}	0.048
Sal (‰)	0.1 ±0.087^{a}	0.12 ±0.067^{a}	0.2 ±0.00^{b}	0.1 ±0.00^{ab}	0.033 ±0.05^{a}	0.000
Eh (mV)	-26.18 ±39^{a}	-52.88 ±75.7^{a}	-19.3 ±2.38^{a}	-106.93 ±59.62^{b}	7.65 ±8.31^{b}	0.000

For each water parameters of a given sites values on a raw affected with different letters in upper case are significantly different at the given probability level $p < 0.05$.

III.1.2.2.2. Heavy metal concentrations in water of lowland sites and applicable norms

The results of the heavy metal concentrations in the lowland waters were presented in Table XVIII. The mean concentrations of Pb, Cd, Cr, Ni, Zn, Cu, As and Co in water during the rainy season was 3.10^{-4}, 2.10^{-3}, 1.10^{-3}, 7.10^{-3}, $1.87.10^{-2}$, $6.7.10^{-3}$, 6.10^{-3} and 3.10^{-4} mg. L^{-1} respectively. During this season, zinc presented the highest value in water, followed by nickel, copper and arsenic. During the dry season, for all heavy metal analysed, their contents in water for each site increased slightly for Pb (1.10^{-2} mg. L^{-1}), Cd ($3.36.10^{-1}$ mg. L^{-1}), Cr ($7.3.10^{-3}$ mg. L^{-1}), Ni ($1.67.10^{-2}$ mg. L^{-1}), Zn (10^{-1} mg. L^{-1}), Cu ($6.3.10^{-3}$ mg. L^{-1}), As ($3.35.10^{-1}$.mg. L^{-1}) and Co ($3.4.10^{-1}$ mg. L^{-1}).

Table XVIII. Heavy metal content in water collected from Mokolo-elobie (site 4), Mvan (site 9) and Atemengue pond Obili (site 11) lowland and permissible limits (mg/L). (RS: Rainy season; DS: Dry season; ^{ns}, ^{a}, ^{ab}, ^{c} and ^{A}, ^{B} statistically not significant, significant among sites and significant between season following Duncan test at $p < 0.05$. * WHO (Anonymous, 2004); ** WWF (Anonymous, 2007); WHO (Anonymous, 2017).

Heavy metals (mg/L)	Rainy season			Dry season			Season effect		Sites effect			Permissible limits (mg/L)
	Site 4	Site 9	Site 11	Site 4	Site 9	Site 11	RS	DS	Site 4	Site 9	Site 11	
Pb	0.001 ±0.0	0.0 ±0.0	0.0 ±0.0	0.011 ±0.0	0.009 ±0.0	0.01 ±0.0	0.0003 ±0.0005^{A}	0.01 ±0.0008^{B}	0.0059± 0.002^{ns}	0.0045 ±0.002^{ns}	0.0052± 0.002^{ns}	0.01*
Cd	0.002 ±0.0	0.002 ±0.0	0.002 ±0.0	0.001 ±0.0	0.001 ±0.0	0.001 ±0.0	0.002 ±0.000^{A}	0.336 ±0.706^{B}	0.0023± 0.000^{ns}	0.0026 ±0.002^{ns}	0.502± 0.34^{ns}	0.003***
Cr	0.001 ±0.0	0.001 ±0.0	0.001 ±0.0	0.008 ±0.006	0.001 ±0.0	0.001 ±0.0	0.001 ±0.000^{A}	0.0073 ±0.0005^{B}	0.0045± 0.002^{ns}	0.0041 ±0.001^{ns}	0.004± 0.001^{ns}	0.005*
Ni	0.011 ±0.0	0.006 ±0.0	0.004 ±0.0	0.014 ±0.0	0.001 ±0.0	0.001 ±0.0	0.007 ±0.003^{A}	0.0167 ±0.002^{B}	0.0128± 0.000^{ns}	0.0127 ±0.003^{ns}	0.01± 0.002^{ns}	0.07*
Zn	0.022 ±0.0	0.023 ±0.0	0.011 ±0.0	0.2 ±0.0	0.001 ±0.0	0.001 ±0.0	0.02 ±0.006^{A}	0.1 ±0.04^{B}	0.0344 ±0.005^{ns}	0.078±0 .025^{ns}	0.065± 0.02^{ns}	3***
Cu	0.007 ±0.0	0.006 ±0.0	0.007 ±0.001	0.001 ±0.0	0.012 ±0.0	0.006 ±0.0	0.0067 ±0.001^{ns}	0.0063 ±0.005^{ns}	0.004± 0.001^{a}	0.009±0 .001^{b}	0.006± 0.0003^{ab}	2***
As	0.011 ±0.0	0.003 ±0.0	0.003 ±0.0	0.001 ±0.0	0.001 ±0.0	0.001 ±0.0	0.0057 ±0.004^{ns}	0.335± 0.7^{ns}	0.006± 0.002^{ns}	0.002±0 .000^{ns}	0.5± 0.34^{ns}	0.01*
Co	0.001 ±0.0	0.0 ±0.0	0.0 ±0.0	0.001 ±0.0	0.001 ±0.0	0.001 ±0.0	0.0003 ±0.0002^{A}	0.34 ±0.2^{B}	0.0038±0. 001^{ns}	0.0035± 0.001^{ns}	0.503± 0.34^{ns}	0.05**

For each metal analysis of a given sites values on a raw affected with different letters in upper case are significantly different at the given probability level $p < 0.05$.

Cadmium showed the highest value in water followed by arsenic, cobalt and zinc. Notably, the heavy metal concentrations in the lowland waters of the Mfoundi watershed followed a decreasing order of As>Co>Cd>Zn>Ni>Cu>Cr>Pb.

The mean concentration of most of the metals in the studied sites was below the WHO standard level, except for Cd, As and Co whose respective concentrations (0.169 mg. L^{-1}), (0.1705 mg. L^{-1}) and (0.1703 mg. L^{-1}) were above the permissible limits for water used for irrigation and agriculture (Anonymous 6, 2007; Anonymous 5, 2017). The mean concentration of cadmium in water was observed to be 0.0023 mg. L^{-1} at site 4 and 0.0026 mg/L at site 9 with the highest being at site 11 (0.502 mg. L^{-1}), thus revealing that its concentrations were slightly above the recommended WHO guideline value in each site studied.

III.1.2.2.4. Multivariate statistical analysis of heavy metals in water

Multivariate statistical analysis including PCA and CA were performed to reveal associations among heavy metals in lowland water and to identify important factors involved in controlling the transport and distribution of metal contaminants. Furthermore, inter-metal interactions may illustrate the sources and pathways of metals present in the particular environments (Proshad et *al*., 2019).

III.1.2.2.4.1. Correlation of physico-chemical characteristics and heavy metals in lowland water during the rainy and dry seasons

In the current study, for the determination of the inter-variable relationships of the studied parameters and the possible sources of the measured pollutant, the Pearson correlation coefficient (r) was applied to the physico-chemical characteristics and heavy metals in the lowland water in both rainy and dry seasons as presented in Fig. 36a&b. The values of the Pearson correlation coefficient ranging between 0.9 and 1 were considered strongly correlated, while values between 0.9 to 0.5 were considered as moderately correlated (Sojobi, 2016; Tirkey et *al*., 2017). The present study also incorporated this classification into the analysis to get an overall idea on the contribution performance of the measured water quality parameters. During the rainy season, it revealed statistically significant ($p<0.01$) positive relationships between EC and Zn ($r = 0.873^{**}$), TDS and Zn ($r = 0.876^{**}$), T and Zn ($r = 0.873^{**}$), Sal and Pb ($r = 0.866^{**}$), Sal and As ($r = 0.866^{**}$), Sal and Ni ($r = 0.971^{**}$), Sal and Co ($r = 0.866^{**}$), Sal and Zn ($r = 0.826^{**}$), Pb and Ni ($r = 0.961^{**}$), As and Ni ($r = 0.961^{**}$) and Ni and Co ($r = 0.961^{**}$). A significant negative relationship was found between pH and Pb ($r = -0.707^{*}$), pH and As ($r = -0.707^{*}$), pH and Co ($r = -0.707$). However, during the dry season, a significant correlation was found between EC and TDS ($r = 0.990^{**}$), Eh and Pb ($r = 0.831^{**}$), Sal and Cr ($r = 0.901^{**}$), Sal and Zn ($r = -0.835^{**}$), Pb and Cu ($r = -0.975^{**}$), Pb and Ni ($r = -0.995^{**}$), Cu and Ni ($r = -0.993^{**}$), Cu and Zn ($r = 0.893^{**}$), Cr and Zn ($r = -0.917^{**}$), and Ni and Zn ($r = 0.834^{**}$).

Other relationships were found to be positive and statistically significant during both rainy and dry seasons between pH and Pb (r = -0.487*), Eh and Cu (r = -0.573*), Pb with Cr, Ni and Zn respectively (r = 0.991**), (r = 0.877**) and (r = 0.280**), Cr with Ni and Zn respectively (r = 0.869**) and (r = 0.775**) and Ni and Zn (r = 0.875**).

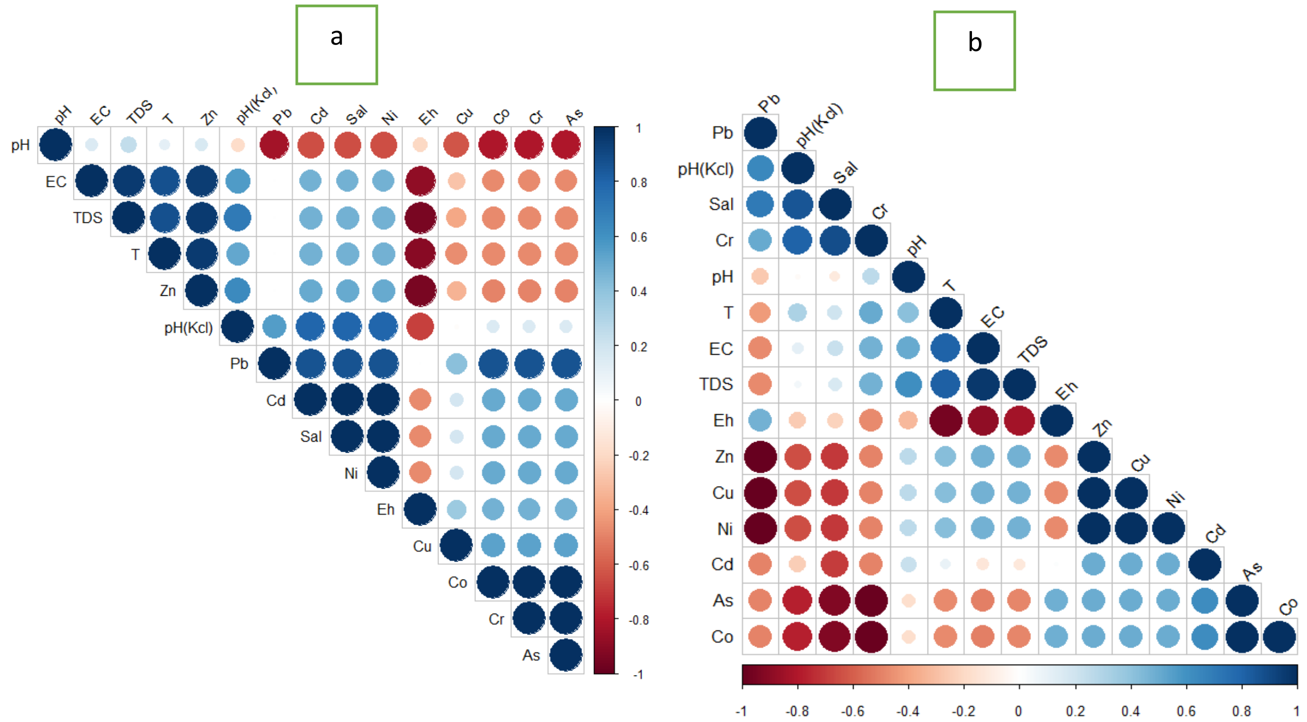

Fig. 36. Correlation coefficient of physicochemical parameters and heavy metals in lowland water (rainy and dry seasons). a. Rainy season, b. Dry season. Significant Pearson correlation ($p < 0.01$) between physico-chemical and heavy metals in lowland water and their significance (both seasons). Positive correlations were displayed in blue and negative correlations in red see (Appendix 7 for all correlation values and their significance). The correlogram represented the correlations for all pairs of variables. The intensity of the color was proportional to the correlation coefficient so the stronger the correlation (i.e., the closer to -1 or 1). The color legend on the right hand side of the correlogram showed the correlation coefficients and the corresponding colors.

III.1.2.2.4.2. Principal component analysis (PCA)

Principal component analysis (PCA) was conducted for the identification of sources of heavy metals in lowland water at sites 4, 9 and 11 during the rainy and dry seasons. PCA is know to be an effective tool for source identification (Islam et *al*., 2018b; Achi et *al*., 2021). The analysis integrated data from eight metal concentrations in the studied lowland waters and explored the possible distribution patterns of metals. The multivariate PCA of heavy metals revealed two major components in the lowland waters accounting for about 92.67% (the rainy season) and 95.82% (the dry season) of the cumulative variance in the data (Fig. 37a&b). In this study, two PCs were computed, and the variances explained by them were 71.24% and 21.43% for water collected during the rainy season in the three sites studied and 53.44% and 42.38% for water collected during the dry season. During the rainy season, Ni, As, Co, and Pb contributed the most (PC1) and were supported by the significant positive correlation Ni (r=0.992**), As (r=0.985**), Co (r=0.985**), Pb (r=0.985**) indicating that the source of the metals in water was mainly from anthropogenic sources (Fig. 37a). The second group (PC2) of the variance with high loads of Cu (r=0.888**) and Zn (r= - 0.669**) showed that the metals in water originated from both anthropogenic and geogenic sources.

Therefore, during the dry season, As, Co and Cd contributed the most in (PC1) and were supported by a significant positive correlations with As (r=0.963**), Co (r=0.963**) and Cd (r=0.962**) which characterized the anthropogenic sources. These elements were mainly from weathering and leaching of these metals into lowland areas, arriving there through direct discharge of untreated/partially treated wastewater from industrial activities or domestic sewage into the lowlands (Dutta et *al.*, 2018). Therefore, Zn, Cu, Ni, Cr and Pb contributed the most in (PC2) and were supported by both significant positive and negative correlations with Zn (r=0.970**), Cu (r=0.945**), Ni (r=0.903**), Cr (r= - 0.836**), Pb (r= - 0.855**), which means that the pollution came from both anthropogenic and natural sources (Fig. 37b).

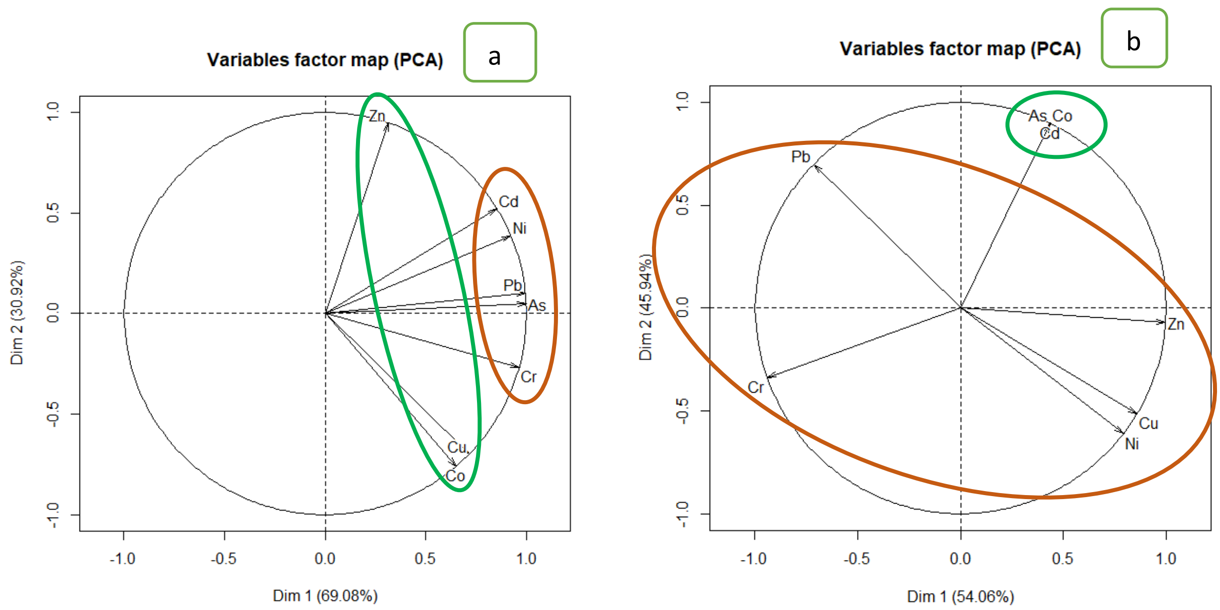

Fig. 37. Principal component analysis (PCA) of heavy metals in lowland water during the rainy and dry seasons. (a. rainy season, b. dry season).

III.1.2.2.4.3. Dendrogram of hierarchical cluster analysis (CA)

Cluster analysis (CA) with dendrogram applying ward's method to divide the similarity and dissimilarity of heavy metals into two main groups is presented in Fig. 38.

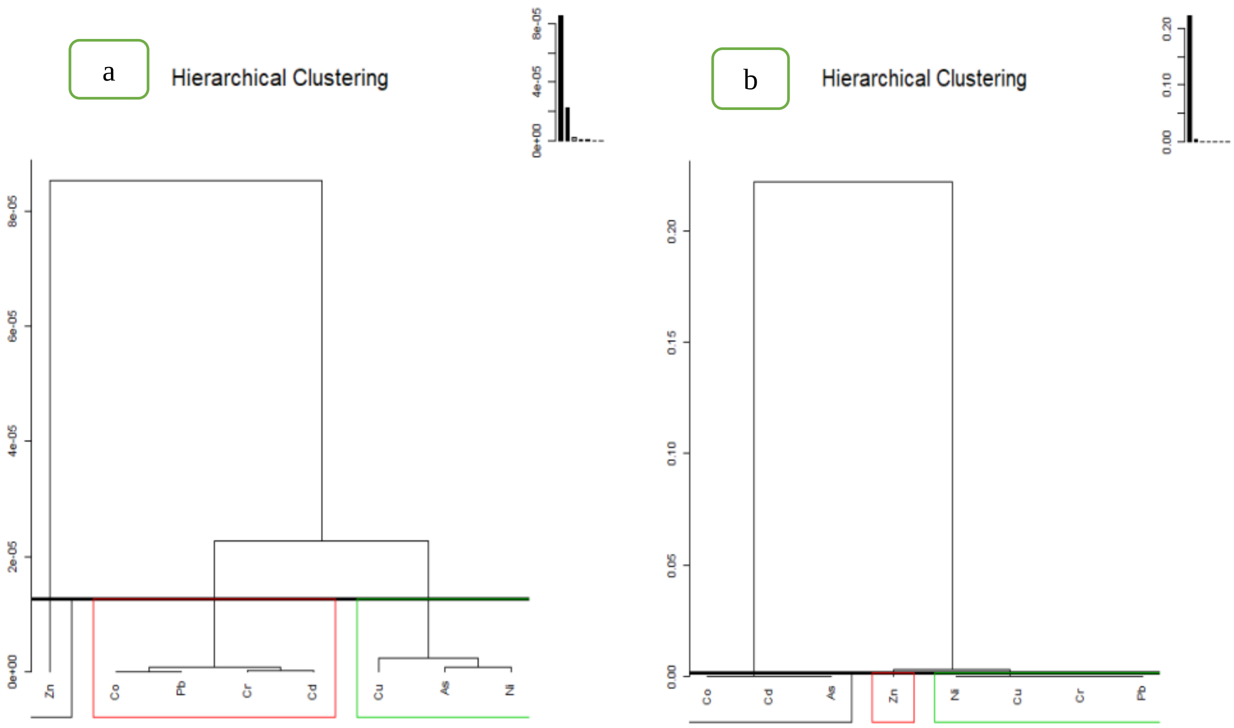

Fig. 38. Dendrogram of hierarchical cluster analysis showing homogeneous group of heavy metals in lowland water samples during the rainy and dry seasons. (a. rainy season, b. dry season).

The groups formed were categorized according to the characteristics of their elements and sources. During the rainy season, Ni, As, Co and Pb formed one cluster where Ni and As were linked to this cluster with a large linkage distance. Cr and Cd were also related to the first cluster, although the Pearson correlation showed nothing for these elements. However, during the dry season, the cluster diagram showed one cluster group formed by As, Co and Cd and the second by Pb, Cr, Cu, Ni and Zn, thus confirming the PCA results.

III.1.2.2.4.4. Sources of heavy metals in different sites during rainy and dry seasons

The PCA of heavy metals revealed two principal components of metals in water collected from sites 4, 9 and 11 during both seasons, accounting for a total of 83.46% of data cumulative variance (Fig. 39). The first principal component (PC1), which contained 60.49% of the calculated variance showed a strong positive loading greater than 0.81 (r >0.81**) for Zn, Pb, Cr, Co and Cd associated with site 11 during the dry season. This showed that the sources of these elements at these sites were primarily anthropogenic during the dry season. However, the second principal component (PC2), which a 22.97% of the measured variance, showed a strong positive loading of Cu at all other sites except site 11_2 (Fig. 39a) during both seasons. This element present in water of all sites in both seasons was asserted to show close anthropogenic and geogenic sources in the environment, which is presented again in the present results with a significant positive loading of r = 0.923**.

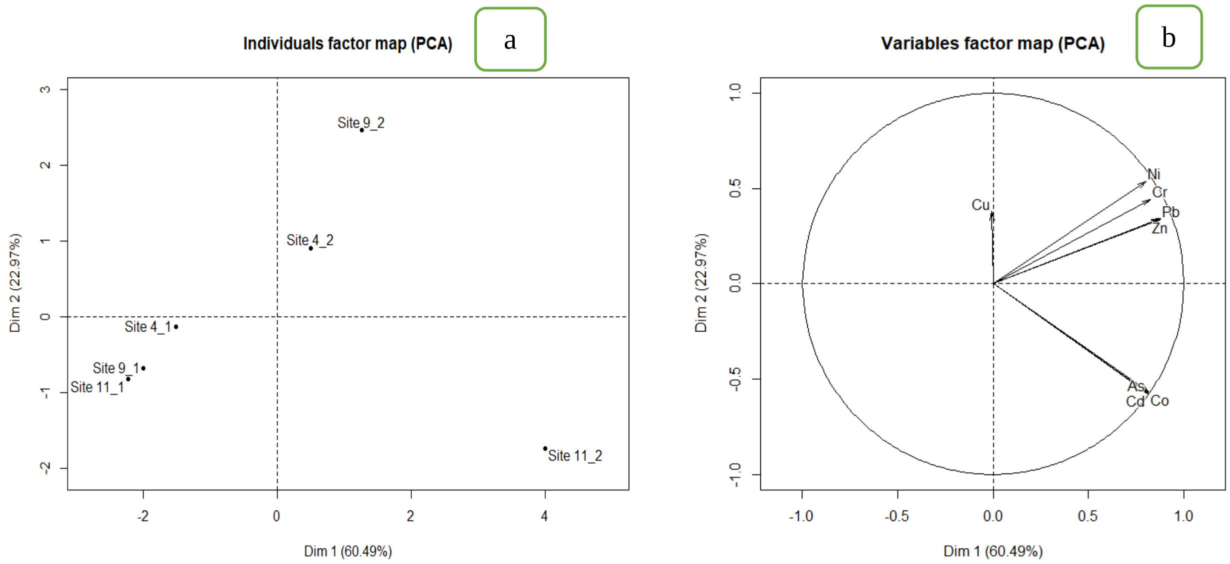

Fig. 39. Sources of heavy metals in the water from different sites during the rainy and dry seasons.

III.1.2.2.5. Assessment of water pollution by heavy metals

III.1.2.2.5.1. Heavy metal toxicity load (HMTL)

The heavy metal toxicity load (HMTL) gives the content in water body and indicates the necessary elimination percentage of toxic metals from water to make it safe for human use. It evaluates the level toxic metals found in water that affects human health. It provided an indication to regulatory authorities about the level of treatment needed to treat river water to acceptable levels for consumption purposes. This index helps to provide an efficient treatment and management plan.

In this study, it determined the pollutant levels of metals in water that resulted in non-carcinogenic risk (Kumar et *al*., 2019). The toxic metals studied as Pb, Cd, Cr, Ni, Zn, Cu, As and Co were selected from the ATSDR substance priority list to calculate HMTL (ATSDR, 2019). The average HMTL was 43116.73 µg/L and ranged from 28737.12 to 60203.14 µg/L during the rainy season. During the dry season, the average HMTL was 141561.94 µg/L and ranged from 85579.49 to 177426.11 µg/L (Table XIX).

The HMTL of the elements in all sites during both seasons was lower than the threshold toxicity load, indicating low contamination of toxic metals in water. During the rainy reason, the total heavy metal toxicity was high for Cd and As, while high for Pb and Cd during the dry season. However, continuous water pollution may lead to an increase in HMTL values in waters of the lowland sites. According to total heavy metal toxicity load, 0.8, 3.00, 8.81 and 39.44% of Co, Cd, As and Zn during the rainy season and 0.32, 1.57, 4.97 and 64.48 % of Pb, Cd, Co and Zn during the dry season respectively need to be removed from water of Yaounde lowland to reduce pollution load.

Table XIX. Heavy metal toxicity load (HMTL) of lowland water during the rainy and dry seasons. ([a] ATSDR, 2019, [b] TMRPL: toxic metal to reduce pollution load, * PTL: within permissible toxicity load, RS: rainy season, DS: dry season).

	Toxicity of heavy metal (µg/L)								
Sampling sites	Pb	Cd	Cr	Ni	Zn	Cu	As	Co	HMTL (µg/L)
				Rainy season (RS)					
Site 4	654.09	2914.5	1330.44	11344.85	19909.99	5788.19	17693.2	567.9	60203.14
Site 9	0.000	2590.5	895.1	6044.57	21147.11	4945.36	4787.27	0.000	40409.92
Site 11	0.000	2342.7	1055.67	3792.95	9831.29	5635.00	5598.11	481.39	28737.12
Total RS	654.09	7847.71	3281.21.	21182.37	50888.4	16368.5	28078.6	1049.29	129350.18
Percent removal of TMRPL (%)	PTL	3.00	PTL	PTL	39.44	PTL	8.81	0.81	
				Dry season (DS)					
Site 4	8712.5	3311.9	7048.42	14543.37	42695.7	812.07	1883.88	6571.65	85579.49
Site 9	7251.77	4196.26	6351.1	19115.75	121672.4	9758.15	2078.45	7002.18	177426.1
Site 11	8406.7	3103.05	6082.8	16006.06	109463	4587.38	6486.42	7544.86	161680.2
Total DS	24370.9	10611.2	19482.3	49665.18	273831.1	15157.6	10448.76	21118.7	424685.83
Percent removal of TMRPL (%)	0.32	1.57	PTL	PTL	64.48	PTL	PTL	4.97	
[a] Hazard intensity score (HIS)	805	1318	893	993	913	805	993	1011	
*Permissible toxicity load (PTL)(µg/L)	23018.1	3964.41	45251.5	70276.5	-	1049900	16685.6	-	

III.1.2.2.5.2. Heavy metal evaluation index (HEI)

The heavy metal evaluation index (HEI) gives the overall water quality with regards to heavy metals. In this study, HEI was evaluated for Pb, Cd, Cr, Ni, Zn, Cu, As, and Co and presented in Table XX. The average HEI values for the rainy and dry seasons were 1.6 and 3.98 respectively with a mean value of 1.61, while the range was 1.23 - 2.36 in the rainy season and 3.96 - 4 in the dry season.

For the lowland waters and in both seasons, 67% of the samples fell below the mean HEI value. In general, the HEI values were classified in terms of pollution levels as low, medium and high with 3.98 as the mean value. According to the HEI classification (Edet & Offiong, 2002), heavy metal pollution in lowland waters was found to be low (less than 10).

Table XX. Heavy metal evaluation index (HEI) of studied metal in lowland sites during the rainy and dry seasons.

	Risk index of single element									
Sampling sites	Pb	Cd	Cr	Ni	Zn	Cu	As	Co	HEI	Risk level
Rainy season										
Site 4	0.08	0.74	0.30	0.16	0.007	0.0004	1.06	0.01	2.36	Low
Site 9	0	0.65	0.2	0.09	0.008	0.0003	0.27	0.000	1.24	Low
Site 11	0	0.6	0.24	0.05	0.004	0.004	0.33	0.01	1.23	Low
Dry season										
Site 4	1.08	0.84	1.57	0.21	0.01	0.0005	0.11	0.13	3.96	Low
Site 9	0.9	1.06	1.42	0.27	0.04	0.006	0.12	0.14	3.97	Low
Site 11	1.04	0.78	0.14	0.23	0.04	0.003	0.38	0.15	4	Low

III.1.2.2.5.3. Ecological risk index (ERI)

The ecological risk index (ERI) evaluated for lowland waters of each site ranged from 26.60 to 48.35 during the rainy season and 35.78 - 81.11 during the dry season season (Table XXI). The average ERI for the rainy and dry seasons was 36.06 and 58.57, respectively. According to the ERI classification reported by Taiwo et *al*. (2020), 100 % of the total samples were found to pose a low ecological risks for both seasons (Fig. 40). Regarding the single element risk index, Cu contributed the most in the rainy season while As and Cu were in the dry season.

Table XXI. Ecological risk index (ERI) of studied metals in lowland sites of Yaounde during the rainy and dry seasons

	Risk index of single element									
Sampling sites	Pb	Cd	Cr	Ni	Zn	Cu	As	Co	ERI	Risk level
Rainy season										
Site 4	10.08	0	5.77	2.4	1.54	22.1	0	6.45	48.35	Low
Site 9	5.80	0	8.23	3.1	1.14	9.81	0	5.13	33.23	Low
Site 11	1.1	0	5.68	3.31	0.75	4.72	0	11.07	26.6	Low
ERI	16.94	0	19.69	8.82	3.44	36.63	0	22.66		
Dry season										
Site 4	5.62	0.1	3.44	1.12	0.96	11.5	8.44	4.58	35.78	Low
Site 9	18.04	0.19	7.73	1.97	1.1	11.33	13.27	5.18	58.82	Low
Site 11	2.48	0.23	16.45	18.27	2.37	7.98	17.09	16.23	81.11	Low
ERI	26.13	0.52	27.63	21.37	4.43	30.81	38.81	26		

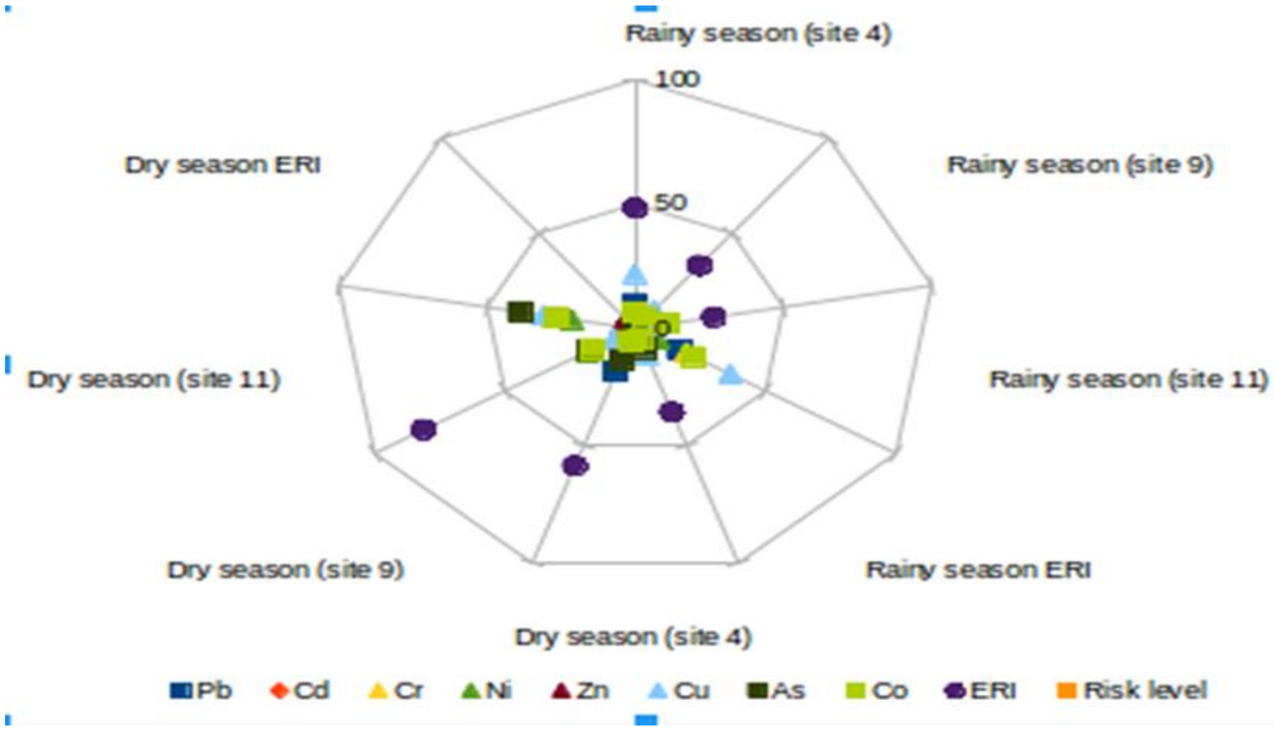

Fig. 40. Ecological risk index (ERI) of studied metal in lowland waters of Yaounde during the rainy and dry seasons.

In this study, the average risk index of a single metal was found in the decreasing order of Cu>Co>Cr>Pb>Ni>Zn during the rainy season. However As and Cd were not identified. Compared to the dry season, the risk index followed the decreasing order of As>Cu>Cr>Pb>Co>Ni>Zn>Cd and As presented the great risk index on the lowland water ecosystem.

III.1.2.2.5.4. Correlation coefficients of metals and pollution indices in lowland water

According to the HTML, HEI and ERI values, 67% of the lowland water samples were below their respective mean values in the rainy season, against 33% in the dry season. These values of HTML, HEI and ERI, which fall below their respective mean values, suggested relatively better water quality as observed by Edet & Offiong (2002). The HTML, HEI and ERI values showed similar trends at the various sampling locations (Fig. 41). In addition, a significant positive correlation ($p<0.05$) was observed between HEI and HMTL values for lowland waters at three sites during both seasons (Table XXII). Upon investigation of the key metals contributing to the computed indices, Cd and As showed significant positive correlations with the measured ERI and HMTL indices, suggesting that these metals were the major contributors to the pollution of Yaounde lowland waters. The pollution of water with Cu is an indication that water is also polluted with other heavy metals. Positive and strong correlation between Co and other metals indicates that the presence of Co reduce the presence of the other metal in water.

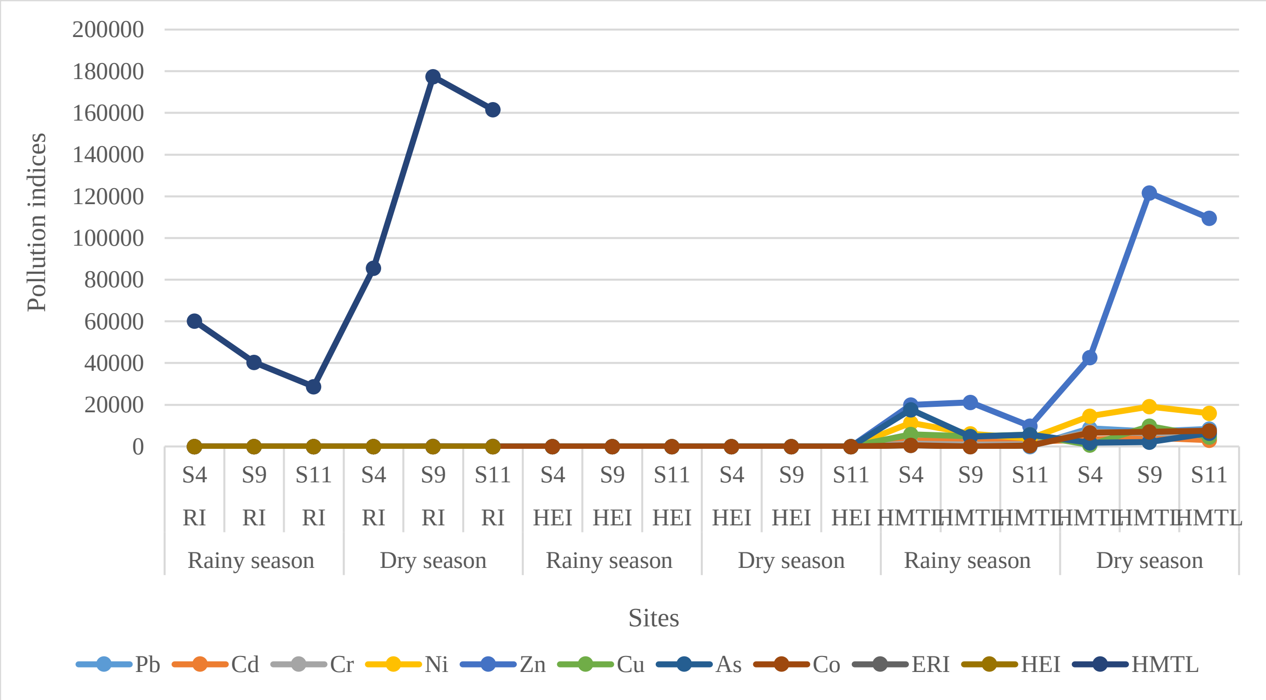

Fig. 41. Spatial distribution of heavy metal pollution indices at different sampling points during the rainy and dry seasons. (ERI: ecological risk index, HTML: heavy metal toxicity load and HEI: heavy metal evaluation index).

On the other hand, Cr and Zn presented a significant positive correlation with ERI only, which means that these two metals were those that posed the main problem in the lowland environment.

Table XXII. Correlation coefficients for metal concentrations and indices values in waters. (ERI: ecological risk index, HTML: heavy metal toxicity load and HEI: heavy metal evaluation index.

	Pb	Cd	Cr	Ni	Zn	Cu	As	Co	ERI	HEI	HMTL
Pb	1										
Cd	0.764**	1									
Cr	0.986**	0.853**	1								
Ni	0.890**	0.953**	0.938**	1							
Zn	0.881**	0.821**	0.894**	0.916**	1						
Cu	0.496*	0.884**	0.606**	0.801**	0.762**	1					
As	0.205	0.639**	0.293	0.558*	0.321	0.639**	1				
Co	0.993**	0.785**	0.984**	0.906**	0.926**	0.565**	0.221	1			
ERI	0.213	0.840*	0.826*	0.785	0.882*	0.091	0.826*	0.557	1		
HEI	0.720	0.637	0.310	0.158	0.424	0.479	0.657	0.210	0.650	1	
HMTL	0.432	0.859*	-0.153	-0.319	-0.069	-0.001	0.895*	-0.263	0.517	0.863*	1

** Correlation is significant at the p = 0.01 level (two-tailed)
* Correlation is significant at the p = 0.05 level (two-tailed).

III.1.2.3. Heavy metal contents in *Echinochloa pyramidalis*, *Pennisetum purpureum* and *Commelina benghalensis* samples collected from lowland in Mokolo-elobi (site 4), Mvan (site 9) and Atemengue pond Obili (site 11) of Yaounde during the rainy and dry seasons.

III.1.2.3.1. Heavy metal concentrations in *E. pyramidalis (Ep)*, *P. purpureum (Pp)* and *C. benghalensis (Cb)* of lowland sites

Heavy metals were analysed in the plant species and the mean concentrations values were reported in Table XXIII. The analysis of the heavy metals in *E. pyramidalis*, *P. purpureum* and *C. benghalensis* during the rainy and dry seasons showed highest concentrations of Pb, Cr, Ni, Zn, Cu and Co in all sites studied (site 4, 9 and 11) with exception to As. These concentrations were above the WHO standard values respectively 0.3 $\mu g.g^{-1}$; 1 $\mu g.g^{-1}$; 0.2 $\mu g.g^{-1}$; 50 $\mu g.g^{-1}$, 10 $\mu g.g^{-1}$ and 0.01 $\mu g.g^{-1}$ for plants (Anonymous 5, 2017; Emurotu & Onianwa, 2017). Among all sites studied and all the metals analysed in plant species, the highest metal concentration was Zn (252.62 $\mu g.g^{-1}$) found in *C. benghalensis* in the dry season. Cadmium (Cd) concentration in *E. pyramidalis* and *P. purpureum* during the rainy season was below 0.2 $\mu g.g^{-1}$, which complies with the standard of Cd in plants (Anonymous 5, 2017), while this level exceeded the limit during the dry season for all species with the maximum value (0.41±0.38) $\mu g.g^{-1}$ in *C. benghalensis*. The average concentration of lead ranged from 1.5 - 9.67 $\mu g.g^{-1}$ in *P. purpureum* and *C. benghalensis* respectively in the dry season.

The chromium concentration in plant species ranged from 11.65 to 22.48 $\mu g.g^{-1}$ in *P. purpureum* in the dry season and *E. pyramidalis* in the rainy season. Ni values varied from 5.57 to 7.79 $\mu g.g^{-1}$ in *P. purpureum* and *E. pyramidalis* in the dry season and rainy seasons respectively.

During the rainy season, among all metals *E. pyramidalis* showed the lowest value of Zn (67.82 µg.g-1), while *C. benghalensis* showed the highest values (252.62 $\mu g.g^{-1}$) during the dry season. The concentration of Cu varied from 19.37 to 56.30 $\mu g.g^{-1}$ respectively in *P. purpureum* and *C. benghalensis* in the dry season and rainy seasons. Therefore, Co content varied from 0.54 to 4.69 $\mu g.g^{-1}$ in *P. purpureum* and *C. benghalensis* respectively and both in dry season.

Table XXIII. Heavy metal concentrations (µg/g) in *E. pyramidalis (Ep)*, *P. purpureum (Pp*) and *C. benghalensis (Cb)* in sites S4, S9, S11 during the rainy and dry seasons and guideline values for plants * (Emurotu & Onianwa, 2017).

Rainy season									
Plant species	Site	Pb	Cd	Cr	Ni	Zn	Cu	As	Co
Ep	S4	3.86±4.92	0±0.00	18.12±6.67	7.75±4.91	89.83±16.38	48.13 ± 11.17	0.00±0.00	1.85±2.61
	S9	3.26 ± 4.61	0.07 ± 0.1	19.87 ± 14.36	6.86 ± 6.55	72.13 ± 2.88	33.61 ± 10.95	0.00±0.00	2.09±2.95
	S11	0.00 ± 0.00	0.5 ± 0.00	29.46 ± 24.75	8.78 ± 6.73	41.50 ± 5.81	5.99 ± 3.32	0.00±0.00	3.46±4.90
Mean±SD		2.37±2.75	0.04±0.056	22.48±9.07	7.79±1.00	67.82±7.10	29.24 ± 4.47	0.00±0.00	2.46±1.23
Pp	S4	4.57±4.03	0.00±0.00	10.26±9.17	5.38±7.58	71.30±19.71	33.81±8.67	0.00±0.00	3.70±5.24
	S9	0.68±0.96	0.07±0.09	21.72±2.39	7.49±0.15	63.65±16.81	21.26±0.07	0.00±0.00	0.40±0.56
	S11	0.00±0.00	0.12±0.02	26.14±25.18	9.04±9.61	134.27±78.97	20.76±3.19	0.00±0.00	1.18±1.67
Mean±SD		1.75±2.10	0.061±0.05	19.37±11.7	7.30±4.98	89.74±35.08	27.28±4.35	0.00±0.00	1.76±2.44
Cb	S4	10.87±3.86	0.11±0.03	23.68±22.47	5.99±5.04	119.54±46.79	94.58±22.49	0.00±0.00	3.09±3.65
	S9	4.03±5.70	0.28±0.31	23.05±20.98	8.25±9.71	111.69±17.69	54.72±22.15	0.00±0.00	4.34±6.13
	S11	1.20±1.70	0.43±0.01	14.72±9.54	5.07±2.83	168.46±125.29	19.60±2.87	0.00±0.00	5.74±7.68
Mean±SD		5.37±2.00	0.27±0.16	20.48±7.07	6.44±3.5	133.23±55.69	56.30±11.23	0.00±0.00	4.39±2.03

Dry season									
Plant species	Site	Pb	Cd	Cr	Ni	Zn	Cu	As	Co
E. pyramidalis	S4	2.38±1.89	0±0.00	15.56±5.33	6.95±2.39	65.83±20.90	32.17 ± 9.93	0.46±0.64	0.61±0.72
	S9	3.93 ± 3.06	0.12 ± 0.17	28.65 ± 11.60	9.22 ± 1.86	196.26 ± 9.70	25.06 ± 3.45	0.00±0.00	3.00±3.49
	S11	2.97 ± 3.18	0.2 ± 0.28	5.56 ± 7.47	2.30 ± 3.16	38.79 ± 15.20	7.84 ± 1.72	0.00±0.00	0.04±0.01
Mean±SD		3.09±0.71	0.11±0.14	16.59±3.19	6.15±0.65	100.29±5.60	21.69±4.33	0.15±0.37	1.22±1.83
P. purpureum	S4	2.33±0.62	0.00±0.00	9.34±2.94	4.80±1.17	89.39±1.51	26.10±5.81	0.00±0.00	0.46±0.52
	S11	0.67±0.84	0.28±0.4	13.97±6.19	6.35±3.94	62.06±7.02	12.64±0.87	0.03±0.05	0.62±0.81
Mean±SD		1.50±0.15	0.14±0.28	11.65±2.29	5.57±1.96	75.73±3.89	19.37±3.49	0.017±0.035	0.54±0.2
C. benghalensis	S4	5.93±1.98	0.13±0.18	8.37±2.26	3.11±0.62	88.45±16.19	32.35±5.14	0.00±0.00	2.12±2.04
	S9	7.17±7.73	0.07±0.09	20.78±11.98	7.05±5.74	143.12±37.95	25.91±2.26	0.00±0.00	4.57±5.42
	S11	1.41±0.47	0.43±0.43	4.40±0.84	2.29±0.83	147.36±35.20	19.51±5.61	0.00±0.00	0.35±0.42
Mean±SD		9.67±6.05	0.41±0.38	22.36±17.09	7.63±5.88	252.62±65.71	25.92±1.82	0.00±0.00	4.69±4.23
Guideline values									
FAO/WHO (2003)		0.3	0.2	0.2	0.2	60	40	-	-
FAO/WHO (2007; 2016)		0.3	0.2	1	-	50	10	0.15	0.01*

Statistical data from ANOVA indicated that metals were accumulated differently by different plant species and in different plant tissues. The statistical analysis presented significant variations between the different sites, seasons, species and tissues of plants, as well as the interaction between plants tissues and plant species (Table XXIV).

Table XXIV. The two-way analysis of variance (ANOVA-2) indicating the difference between the investigated heavy metals in the different tissues of the three plant species collected from three different lowland sites in Yaounde.

Tests	Df	Heavy metals							
		Pb	Cd	Cr	Ni	Zn	Cu	As	Co
Fsites	3	11.91***	13.89***	4.43*	2.22^{ns}	1.94^{ns}	12.82***	1.11^{ns}	1.13^{ns}
Fspecies	3	3.47*	0.91^{ns}	0.61^{ns}	1.13^{ns}	3.33*	3.46*	2.94^{ns}	4.71*
Fseasons	2	1.17^{ns}	11.1**	5.15*	3.21^{ns}	0.049^{ns}	5.36*	39.87***	1.47^{ns}
Fplant tissues	3	13.80***	7.28**	31.35***	41.77***	5.47**	6.77**	12.18***	35.4***
Fplant tissues × plant species	6	4.22**	8.9***	1.03^{ns}	0.57^{ns}	0.72^{ns}	1.18^{ns}	0.8^{ns}	0.76^{ns}

(df degree of freedom, * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$, and ^{ns} not significant (i.e. $p > 0.05$)).

III.1.2.3.2. Influence of environmental factors on heavy metal concentrations in *E. pyramidalis (Ep)*, *P. purpureum (Pp)* and *C. benghalensis (Cb)* in lowland sites

III.1.2.3.2.1. Effects of plant species on metal accumulation

Analysis of heavy metals in each of the plant species (*E. pyramidalis*, *P. purpureum* and *C. benghalensis*) indicated that among all metals, only Zn showed greater variation in the investigated species. Data analysis presented a significant difference ($p < 0.05$) for Zn accumulation in plants. *C. benghalensis* and *P. purpureum* accumulated respectively 1.5 and 1 times higher concentrations of Zn than *E. pyramidalis* (Fig. 42). Among the three plant species, *C. benghalensis* showed the highest accumulation of Zn, followed by *P. purpureum*. The decreasing order of accumulation is presented as *Cb*>*Pp*>*Ep*.

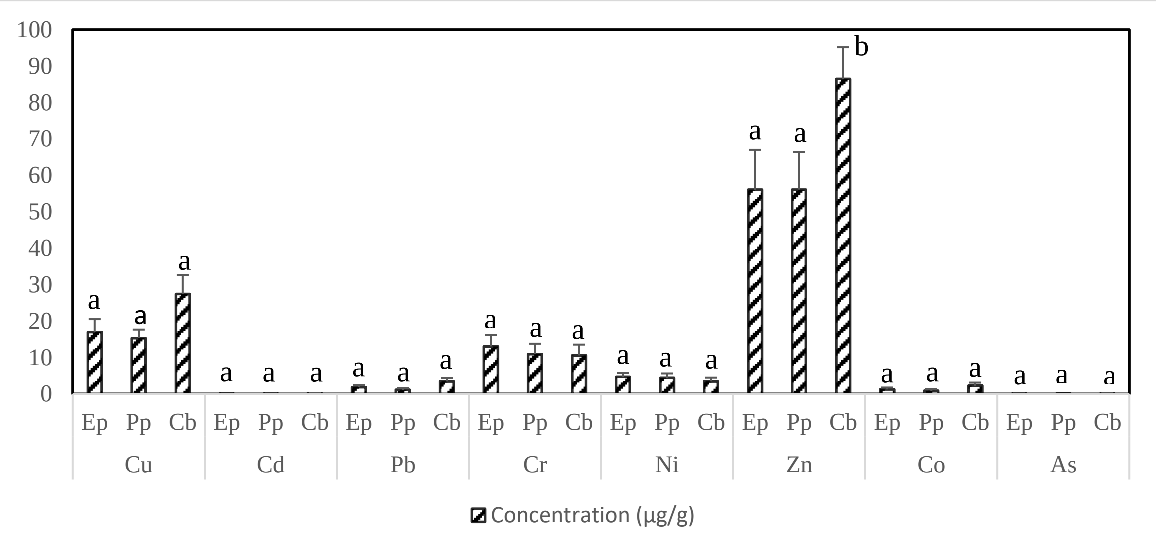

Fig. 42. Variation of the metal concentrations in *E. pyramidalis (Ep)*, *P. purpureum (Pp)* and *C. benghalensis (Cb)*. (For each metal, mean values followed by the same letter are not significantly different following Duncan test at $p < 0.05$).

III.1.2.3.2.2. Plant tissue effects on metal accumulation

The results showed that the constitution of each plant tissue or organ (roots, stems and leaves) of the three plants affected significantly the accumulation of all the heavy metals studied (Pb, Cd, Cr, Ni, Zn, Cu, As, Co) with the exception of As where p (0.16) was higher than the p-value (0.05). Cd and Zn were accumulated differently in plant organs than the other metals. For Cu, Cr, Ni, Pb and Co, roots followed by leaves accumulated respectively 2.5 and 1.4 times, 6 and 1.5 times, 6.5 and 1.6 times, 11.4 and 2.3 times and 148 and 5 times higher levels of theses metals compared to stems. For Zn accumulation, roots concentrated around 95% and stems 25% compared to leaves (Fig. 43).

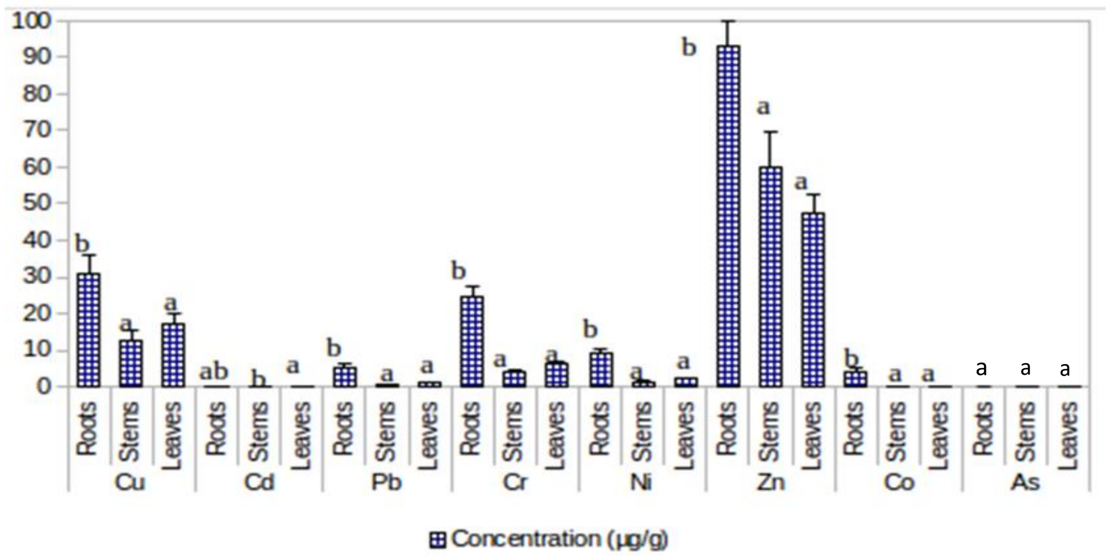

Fig. 43. Variation of the metal contents in roots, stems and leaves. (For each plant tissue, mean values of metals followed by the same letter are not significantly different following Duncan test at $p < 0.05$).

Contrastly, Cd was more concentrated in stems and roots, 338% and 111% respectively as compared to Cd in leaves. This study showed that roots accumulated more metals than leaves and stems. The decreasing order of metals in plant tissues is presented as follows: roots>leaves>stems. Fig. 44 showed the root systems and the other parts of the three plant species studied. However, variability was found amount the Zn and Cd accumulation in the plant tissues, which was preferentially done in roots.

Fig. 44. Root systems and other parts of *C. benghalensis*, *P. purpureum* and *E. pyramidalis*. (a. leaves, b. stems and c. roots of *C.benghalensis*; d. leaves, e. stems and f. roots of *P. purpureum*; g. leaves, h. stems and i. roots of *E. pyramidalis*).

III.1.2.3.2.3. Season effects on the metal contents in plants

In this study, the seasons (rainy and dry) affected significantly the concentrations of Pb, Cu, and As in plants. During the rainy season, the concentration of Pb and Zn were 1.1 times higher than their contents during the dry season. In addition, As concentration was less than one time higher in the rainy season than in the dry season (Fig. 45).

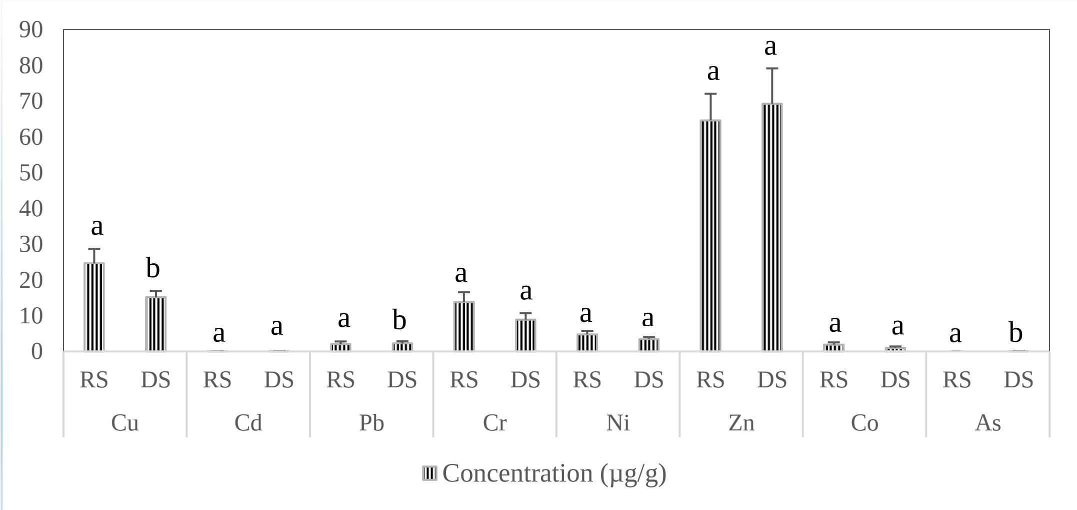

Fig. 45. Variation of the metals contents in plants during rainy and dry seasons (RS, DS). (For each season, mean values of metals followed by the same letter are not significantly different following Duncan test at $p<0.05$).

III.1.2.3.2.4. Sites effects on metal concentrations in plants

Considering the effects of sites, the levels of Cu, Cr, Pb, Ni and Co in plants were significantly affected by the different sites (site 4, 9 and 11). The concentrations of Cr and Ni in plants were 2.4 and 1.7 times and 6.1 and 1.4 times higher in sites 11 and 9 respectively than these metals in site 4. Therefore, the concentration of Cu in plants was 2.6 and 1.7 times higher in site 4 and 9 respectively than Cu in site 11. Pb content was about 7 and 4.4 times higher in sites 9 and 4 respectively than Pb in the same site 11. Inversely, the content of Co in plants was 165% and 7% higher in sites 11 and 4 than Co in site 9. Thus, the constituents of site 4 were most favourable for the uptake of heavy metals by plants (Fig. 46).

III.1.2.3.2.5. Effects of plant tissues and plant specie interactions on heavy metal contents in plant species

The analysis of heavy metals in the different sites and plant tissues showed that for all metals studied, the interception of plant tissues and plant species was statistically not significant for the accumulation of all metals where p (0.606) was higher than 0.05.

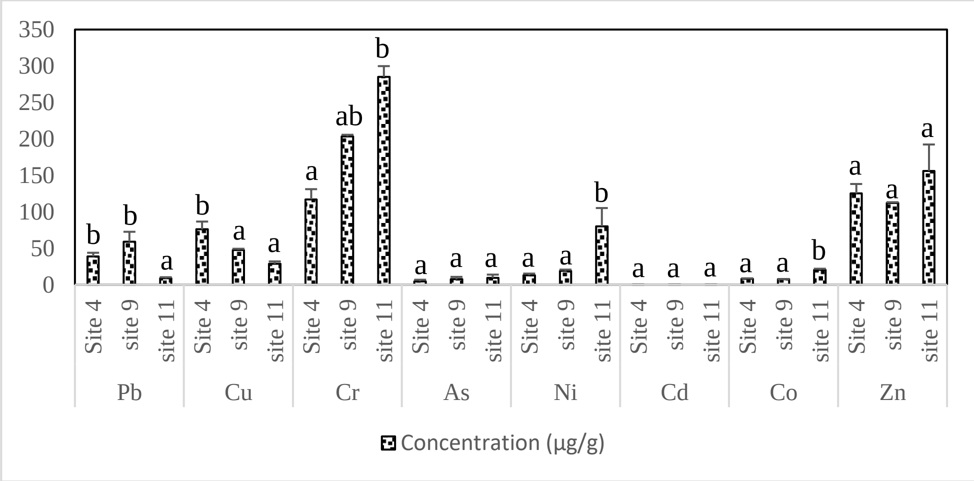

Fig. 46. Variation of metal contents in plants in different sites (site 4, 9 and 11). For each site, mean values of metals followed by the same letter are not significantly different following Duncan test at p<0.05).

In general, the results obtained indicated that for Cu, Pb, Zn and Co, the roots of *C. benghalensis* accumulated the highest concentration of these metals compared to shoots (stems + leaves) of *P. purpureum* (Fig. 47). Differently, Cr and Ni were also accumulated in the roots, but their concentrations were higher in *E. pyramidalis* and *P. purpureum* respectively than their contents in the stems of *C. benghalensis*. Contrarily, Cd was more accumulated by stem of *C. benghalensis* than roots of *E. pyramidalis*.

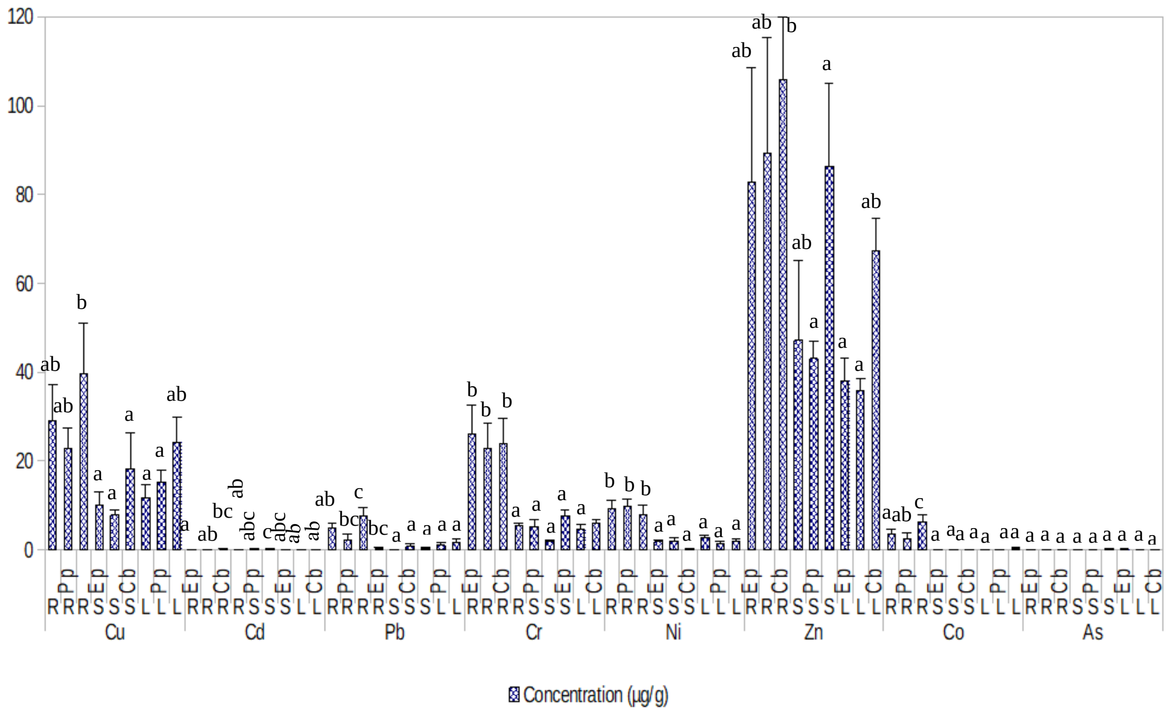

Fig. 47. Plant tissue interactions with plant species on metal accumulations. (R: roots, S: stems, L: leaves, *Ep: E. pyramidalis*, *Cb: C. benghalensis*, *Pp: P. purpureum*). For each interaction plants-plant tissues and mean values of metals followed by the same letter are not significantly different following Duncan test at p<0.05).

III.1.2.4. Correlation between heavy metal contents in plant tissues and other matrices (soils and water)

The inter-metal relationships provided interesting information on the sources and pathways of metals (Wang et *al*., 2017; Zwolak & Sarzy, 2019). Inter-metal relationships support the results obtained from PCA, and the correlation matrix (CM) has been useful in revealing some new associations of metals that were not properly stated in the previous analysis. The inter-relationship between heavy metals in plant tissues, soils and waters was summarized in Table XXV. Heavy metal analysis showed that some pairs have strong significant correlations ($P < 0.01$) between *E. pyramidalis* tissues, soil and water such as Pb-Cu (r = 0.685**), Cu-Pb (r = 0.798**), Cr-Ni (r = 0.887**), Cr-Zn (r = 0.817**), Cr-Co (r = 0.905**), Ni-Cr (r = 0.971**), Zn-Cr (r = 0.689**), Co-Cr (r = 0.914**) and Zn-Cu (r = 0.539**), Zn-Co (r = 0.760**), Cu-Zn (r = 0.656**) and Co-Zn (r = 0.546**). Similarly, a strong correlation was also observed in *P. purpureum* parts, soils and waters Pb-Cu (r = 0.708**), Cu-Pb (r = 0.815**), Cr-Ni (r = 0.887**), Cr-Zn (r = 0.859**), Cr-Co (r = 0.900**), Ni-Cr (r = 0.966**), Ni-Zn (r = 0.817**), Zn-Ni (r = 0.863**),Zn-Cu (r = 0.525**), Cu-Zn (r = 0.678**). In the organs of *C. benghalensis*, waters and soils for Pb-Cu (r = 0.587**), Cu-Pb (r = 0.796**), Cr-Ni (r = 0.891**), Cr-Zn (r = 0.756**), Cr-Co (r = 0.866**), Ni-Cr (r = 0.970**), Zn-Cr (r = 0.565**), Co-Cr (r = 0.863**), Ni-Zn (r = 0.800**), Zn-Ni (r = 0.575**), Ni-Co (r = 0.847**) and Co-Ni (r = 0.886**).

In this study, Cd and As were the two metals which presented weak inter-relationship with other metals in the three plant species, in soils and in water during both seasons.

III.1.3. Remediation performance of plant species

III.1.3.1. Removal efficiency of metals by plants

III.1.3.1.1. Heavy Metal Transfer from roots to shoots

The translocation or transfer factor (TF) primarily assessed the phytoextractive capacities of plant species. In the present study, TF values were displayed for the three sites during the rainy and dry seasons. The TF values varied for Pb (0 to 0.6), Cd (0 to 1.29), Cr (0.19 to 0.86), Ni (0 to 0.43), Zn (0.41 to 3.22), Cu (0.44 to 1.40), As (0) and Co (0 to 0.1) during the rainy season. During the dry season, these TF values were (0 to 15.99) Pb, (0 to 5.99) Cd, (0.22 to 38.58) Cr, (0.1 to 70) Ni, (0.63 to 1.77) Zn, (0.55 to 1.51) Cu, (0) As and Co (0 to 0.59). During the rainy season, the highest transfer values were observed for zinc in all sites, with exception of site 11 for *P. purpureum* compared to those of the other metals. The translocation factor of *C. benghalensis* for zinc during the rainy season was higher at all three sites, with a maximum value of 3.22 at site 11, followed by 1.77 at site 4 and 1.25 at site 9 (Fig. 48). *P. purpureum* showed a TF value higher than 1 in sites 4 (1.49) and 9 (1.46) followed by *E. pyramidalis* in site 4.

Table XXV. Pearson correlation coefficient (r-value) between heavy metals in the soils, waters and three plant specie tissues in the lowlands of Yaounde. (*Ep: E. pyramidalis, Pp: P. purpureum* and *Cb: C. benghalensis*, *. Correlation is significant at the 0.05 level (2-tailed), **. Correlation is significant at the 0.01 level (2-tailed)).

	Soil \ Water	Pb	Cd	Cr	Ni	Zn	Cu	As	Co
Ep parts	Pb	1	-0.122	0.596**	0.694**	0.645**	0.798**	-0.126	0.580**
	Cd	-0.233	1	-0.130	-0.136	-0.111	-0.154	0.903**	0.124
	Cr	0.412*	-0.234	1	0.971**	0.689**	0.652**	-0.145	0.914**
	Ni	0.024	-0.117	0.887**	1	0.689**	0.785**	-0.144	0.868**
	Zn	0.350*	-0.170	0.817**	0.771**	1	0.656**	-0.143	0.546**
	Cu	0.684**	-0.362*	0.399*	0.142	0.539**	1	-0.134	0.509**
	As	0.492**	-0.114	0.752**	0.707**	0.612**	0.225	1	0.100
	Co	0.223	-0.261	0.905**	0.854**	0.760**	0.342*	0.620**	1
	Soil \ Water	Pb	Cd	Cr	Ni	Zn	Cu	As	Co
Pp parts	Pb	1	-0.095	0.276	0.438*	0.263	0.815**	-0.088	0.805**
	Cd	-0.251	1	-0.093	-0.101	-0.111	-0.162	0.985**	0.199
	Cr	0.393*	-0.259	1	0.966**	0.897**	0.632**	-0.133	0.490**
	Ni	0.003	-0.128	0.887**	1	0.863**	0.705**	-0.133	0.647**
	Zn	0.329	-0.229	0.859**	0.817**	1	0.678**	-0.162	0.386*
	Cu	0.708**	-0.369*	0.403*	0.125	0.525**	1	-0.178	0.650**
	As	0.480**	-0.137	0.746**	0.702**	0.639**	0.226	1	0.209
	Co	0.209	-0.290	0.900**	0.849**	0.768**	0.355*	0.616**	1
	Soil \ Water	Pb	Cd	Cr	Ni	Zn	Cu	As	Co
Cb parts	Pb	1	-0.044	0.878**	0.843**	0.568**	0.796**	-0.102	0.711**
	Cd	-0.327	1	0.034	0.047	0.043	-0.023	0.818**	0.209
	Cr	0.402*	-0.368*	1	0.970**	0.565**	0.779**	-0.137	0.863**
	Ni	0.025	-0.203	0.891**	1	0.575**	0.725**	-0.141	0.886**
	Zn	0.219	-0.072	0.756**	0.800**	1	0.571**	-0.164	0.469**
	Cu	0.587**	-0.262	0.267	0.067	0.277	1	-0.130	0.525**
	As	0.482**	-0.242	0.751**	0.710**	0.585**	0.115	1	-0.005
	Co	0.180	-0.246	0.866**	0.847**	0.672**	0.212	0.586**	1

The results analysed showed that, zinc was the most extracted metal among all others. *C. benghalensis* and *P. purpureum* exhibited high values of Cu and Cd transfer (TF>1), while Ni was transferred only by *P. purpureum* during both seasons. The results of the present study showed that *C. benghalensis* could be used mainly as phytoextractor for zinc in the rainy season specifically. It could also be used as phytoextractor for Cu and Cd, while *P. purpureum* can be used for Cu-Cd-Ni not with great performance.

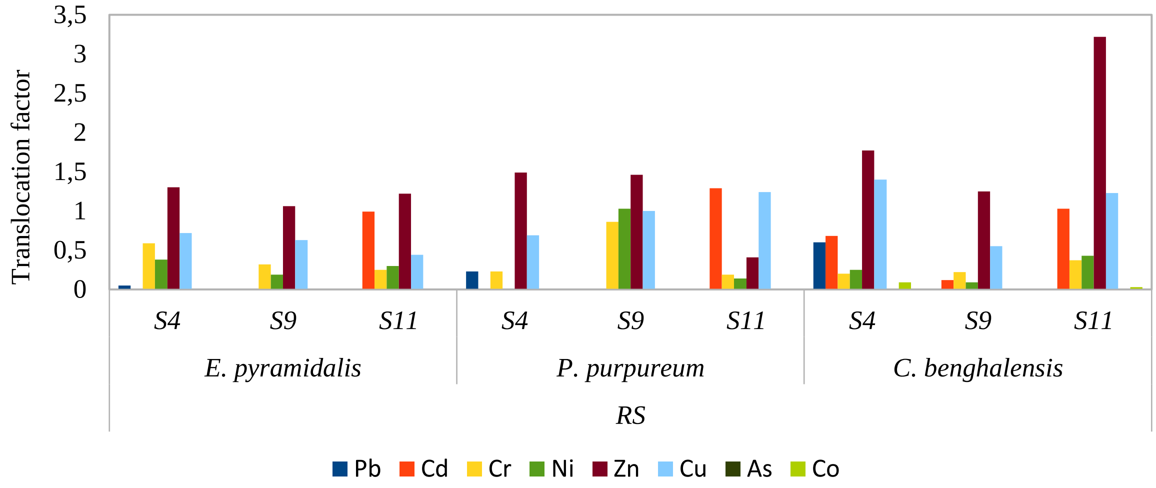

Fig. 48. Translocation factor of heavy metals in plant species growing in lowlands during the rainy season. (S4: (site 4) Mokolo-elobi, S9: (site 9) Mvan and S11: (site 11) Atemengue pond Obili, RS: rainy season.

During the dry season, the transfer value of nickel (70) in *E. pyramidalis* of site 11 was the highest, followed by chromium (38.58) compared to other metals in all sites. In addition, in site 11 *P. purpureum* showed the highest transfer value of 15.99 for lead while *C. benghalensis* presented the highest transfer value (5.99) for cadmium (Fig. 49).

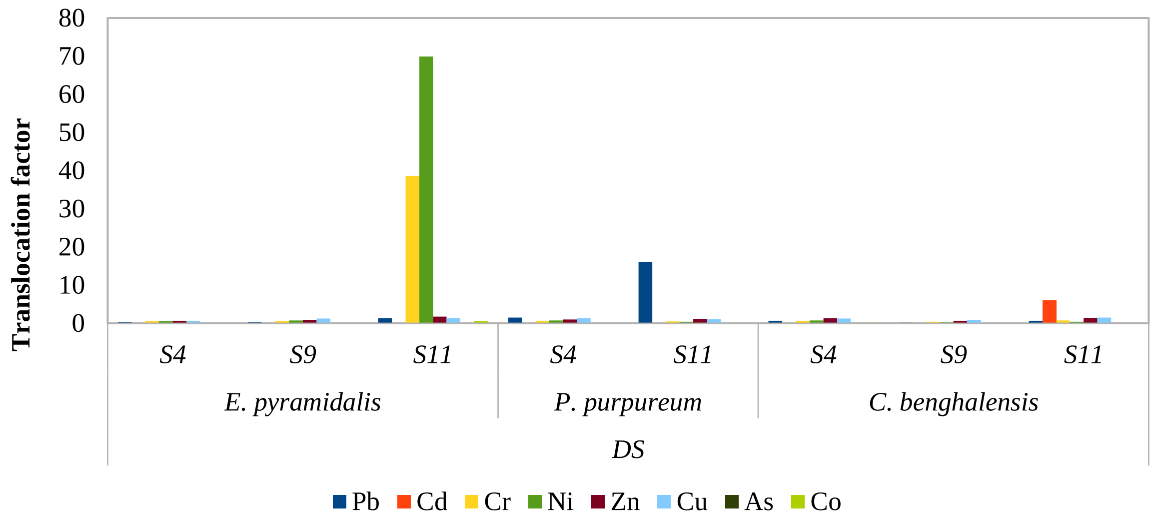

Fig. 49. Translocation factor of heavy metals in plant species growing in lowlands during the dry season. (S4: (site 4) Mokolo-elobi, S9: (site 9) Mvan and S11: (site 11) Atemengue pond Obili, DS: dry season.

This study revealed the influence of seasons in the transfer of metals by plants. Therefore, in relation to the rainy season, each plant was considered as a specific metal extractor. Thus, *E. pyramidalis* can be used as a phytoextrator for Ni and Cr, while *P. purpureum* can be used as a phytoextrator for Pb in the lowland areas.

III.1.3.1.2. Metal mobility from soil to shoot

The values of the mobility ratio in shoots in all lowland sites were showed in Table XXVI. Of all metals analyzed, all three species were enriched in Cd, Zn and Cu independently of seasons. The values of the mobility ratio for Pb, Cr, Ni, As and Co in the shoots of the three plant species from contaminated sites showed that their absorption from the soil was not considerable (MR<1).

Table XXVI. Mobility ratio (MR) of metals in three plant species. (*Ep*: *E. pyramidalis*, *Pp*: *P. purpureum* and *Cb*: *C. benghalensis*, S4: (site 4) Mokolo-elobi, S9: (site 9) Mvan and S11: (site 11) Atemengue pond Obili, RS: rainy season, DS: dry season).

			MR							
Season	Plants species	Sites	Pb	Cd	Cr	Ni	Zn	Cu	As	Co
RS	*E. pyramidalis*	S4	0.008	0	0.1	0.24	0.66	0.40	0	0
	(*Ep*)	S9	0	0	0.047	0.1	0.65	0.58	0	0
		S11	0	2.74	0.08	0.03	0.19	0.10	0	0
	P. purpureum	S4	0.034	0	0.02	0.001	0.55	0.28	0	0
	(*Pp*)	S9	0	0	0.10	0.33	0.66	0.48	0	0
		S11	0	7	0.18	0.07	0.57	0.64	0	0.05
	C. benghalensis	S4	0.22	0	0.16	0.33	0.77	0.95	0	0.32
	(*Cb*)	S9	0.14	0	0.11	0.36	0.97	1.23	0	0.57
		S11	0	24.42	0.02	0.022	1.08	0.6	0	0.013
DS	*E. pyramidalis*	S4	0.04	0	0.14	0.63	0.53	0.48	0.1	0.01
	(*Ep*)	S9	0.02	16.23	0.1	0.54	1.72	0.54	0	0.07
		S11	0.14	0	0.025	0.18	0.67	0.42	0	0.002
	P. purpureum	S4	0.1	0	0.08	0.47	0.92	0.58	0	0.01
	(*Pp*)	S11	0.24	0	0.02	0.14	0.89	0.62	0	0.003
	C. benghalensis	S4	0.16	31.15	0.08	0.38	1.04	0.7	0	0.1
	(*Cb*)	S9	0.02	9	0.06	0.20	1.06	0.47	0	0.1
		S11	0.2	0	0.009	0.028	2.28	1.1	0	0.003

However, a very high mobility ration (MR) for Cd in *E. pyramidalis* (2.74), *P. purpureum* (7) and *C. benghalensis* (24.42) at site 11 was observed in the rainy season, similarly for Cu (1.23) and Zn (1.08) in *C. benghalensis*.

During the dry season, *C. benghalensis* and *E. pyramidalis* showed higher MR values for Cd uptake, 31.15 and 16.23 for sites 4 and 9 respectively. The MR values of *E. pyramidalis* and *C. benghalensis* for Zn uptake were higher than 1 (MR>1). Only *C. benghalensis* in site 11 uptaked Cu among all the other sites studied. This study revealed that, *E. pyramidalis*, *P. purpureum* and *C.*

benghalensis were Cd-accumulators in lowland sites during both seasons. *C. benghalensis* can also be used in the phytoaccumulation of Zn and Cu in the lowlands.

III.1.3.1.3. Metal bioaccumulation capacities in roots, stems and leaves

The bioaccumulation factors (BAF) analyzed in leaves, stems, and roots were presented in Table XXVII. In all the three studied sites, Cd showed the highest BAF value of 31.15. During the rainy season, a high range of BAF value of *E. pyramidalis* was observed in site 11 in roots and stems for Cd (2.768 and 1.956) respectively. Cd in *P. purpureum* was shown higher in roots (5.754) and leaves (3.795). However, BAF in *C. benghalensis* showed high values in site 11 in roots and stems (23.78 and 24.29) respectively. In addition, Cu and Co in site 9 were higher than 1.

Comparatively during the dry season, Cd was found higher in stems for *E. pyramidalis* in site 9 (16.232) while Ni and Zn were higher than 1 respectively in site 4 and 9. *P. purpureum* in all sites was lower than 1. Therefore, Cd was very high in *C. benghalensis* of sites 11 (31.154) and 9 (8.991). The highest bioaccumulation of Cd was noticed in the stems of *C. benghalensis*.

III.1.3.2. Metal bioaccumualation efficiency

III.1.3.2.1. Metal accumulation index

The MAI values for the roots and shoots (leaves + stems) were summarized in Table XXVII. The metal accumulation index gave the overall performance of plant species to accumulate metals based on its deviation in metal uptake. The results of the present study showed different variations in plant species, sites, seasons and plant tissues for metal uptake. In all the studied sites, the highest value of MAI was observed in shoots of *P. purpureum* at site 9 (47.73), and the lowest was found in shoots of *P. purpureum* collected at site 4 (1.04) during the rainy season. However, during the dry season, *P. purpureum* showed the highest value in roots (9.60) in site 4, while the lowest value was found in the roots of *E. pyramidalis* in site 4 (1.36). Among the three plant species, 82% of the metals were accumulated in the roots, while 18 % were in the shoots. The average MAI could be arranged in the rainy and dry seasons in the following decreasing order: *P. purpureum* > *E. pyramidalis* > *C. Benghalensis*.

III.1.3.2.2. Comprehensive bioconcentration index

The present study compared native plant species from three different lowland sites growing under their natural environmental conditions. The comprehensive bioconcentration index (CBCI index) revealed the overall performance of plants in terms of bioaccumulation of several metals to assess their phytoremediation capacities (Zhao et *al*., 2014). Applying the CBCI to data for each specie studied in each site during the rainy and dry seasons, *P. purpureum* in site 11 showed the highest CBCI value (43.5) during both seasons. Higher CBCI values were also found in the following plant species during each season: *E. pyramidalis* (32.37) in site 11, *C. benghalensis* (19.75) in site 4 during the rainy season, *P. purpureum* (18.49) in site 11 dry season, *C. benghalensis* (11.97) at site 11 and *P. purpureum* (5.58) at site 4 during the rainy season. All the values of CBCI for these species in the specific sites and seasons were greater than 5 (CBCI > 5), meaning that, these native plants have the exceptional ability to accumulate various metals simultaneously.

Hierarchical cluster analysis (HCA) was applied to the CBCI values of plants at all three sites during both seasons (Fig. 50). In this figure, group 1 was represented by the green color, group 2 by the red color and group 3 by the black color. The analysis reduced the variables into three groups, explaining that *E. pyramidalis* and *P. purpureum* were associated with high CBCI values for plant shoots in site 11 during the rainy season (group 1). However, *C. benghalensis* and *P.purpureum* displayed a strong association with CBCI values in sites 4 and 11 respectively during the dry season. The same was observed in site 11 during the rainy season for *C. benghalensis* (group 2). In group 3, almost all CBCI values of the three plants in the three sites were associated with both seasons. In this group, high variability was observed in *E. pyramidalis*, *P. purpureum* and *C. benghalensis* across the sites and seasons. Furthermore, as shown in Table XXVII, the CBCI results indicated that the Poaceae families exhibited a higher capacity for accumulation of different metals as compared to Commelinaceae. *E. pyramidalis* in site 11 in the rainy season (32.37), *P. purpureum* in site 11 in the rainy season (43.5) and in the dry season (18.49) and *C. benghalensis* in site 11 in the rainy season (11.97) and site 4 in the dry season (19.75).

Table XXVII. Bioaccumulation factor (BAF), metal accumulation index (MAI) and comprehensive bioconcentration index (CBCI) of roots and shoots of *E. pyramidalis (Ep), P. purpureum (Pp)* and *C. benghalensis (Cb)* in lowland sites of Mokolo-elobi (S4), Mvan (S9) and Atemengue pond Obili (S11) in the rainy and dry seasons.

					BAF								MAI	CBCI
Season	Plant species	Sites	Plant tissues		Pb	Cd	Cr	Ni	Zn	Cu	As	Co		
RS	*E. pyramidalis (Ep)*	S4		Roots	0.147	0	0.154	0.627	0.507	0.561	0	0.386	2.30	4.84
		S4	Shoots	Stems	0	0	0.039	0.081	0.335	0.185	0	0	1.59	
				Leaves	0.008	0	0.122	0.433	0.322	0.217	0	0		
		S9		Roots	0.227	0	0.144	0.498	0.612	0.307	0	0.548	4.35	6.30
		S9	Shoots	Stems	0	0	0.020	0.031	0.329	0.276	0	0	3.82	
				Leaves	0	0	0.027	0.066	0.319	0.932	0	0		
		S11		Roots	0	2.768	0.324	0.1	0.158	0.231	0	0.288	19.20	32.37
		S11	Shoots	Stems	0	1.926	0.026	0.009	0.096	0.008	0	0	18.49	
				Leaves	0	0.814	0.056	0.021	0.096	0.092	0	0		
DS	*E. pyramidalis (Ep)*	S4		Roots	0.134	0	0.225	1.034	0.84	0.754	0	0.09	2.32	1.27
		S4	Shoots	Stems	0.001	0	0.044	0.212	0.304	0.190	0.1	0.165	1.43	
				Leaves	0.036	0	0.093	0.417	0.227	0.754	0	0.008		
		S9		Roots	0.068	0	0.185	0.716	1.848	0.442	0	0.772	4.99	1.78
		S9	Shoots	Stems	0.007	16.232	0.035	0.200	1.210	0.222	0	0.013	4.46	
				Leaves	0.013	0	0.068	0.337	0.513	0.315	0	0.056		
		S11		Roots	0.31	0	0.0007	0.003	0.372	0.310	0	0.003	1.36	4.075
		S11	Shoots	Stems	0.13	0	0.008	0.064	0.213	0.132	0	0.0006	1.89	
				Leaves	0.29	0	0.017	0.12	0.444	0.292	0	0.001		

								BAF					MAI	CBCI
Season	Plant species	Sites	Plant tissues	Pb	Cd	Cr	Ni	Zn	Cu	As	Co			
		S4		Roots	0.149	0	0.113	0.60	0.372	0.399	0	0.775	1.75	2.68
RS	*P. purpureum (Pp)*	S4	Shoots	Stems	0	0	0.014	0.0013	0.288	0.079	0	0	1.04	
				Leaves	0.034	0	0.011	0	0.264	0.198	0	0		
		S9		Roots	0.047	0	0.112	0.320	0.452	0.478	0	0.104	47.55	1.66
		S9	Shoots	Stems	0	0	0.057	0.2	0.290	0.181	0	0	47.73	
				Leaves	0	0	0.04	0.13	0.369	0.299	0	0		
		S11		Roots	0	5.754	0.303	0.116	0.802	0.513	0	0.098	2.23	43.5
		S11	Shoots	Stems	0	3.638	0.028	0.009	0.212	0.323	0	0	1.88	
				Leaves	0	3.795	0.029	0.008	0.119	0.315	0	0		
		S4		Roots	0.068	0	0.133	0.673	0.943	0.423	0	0.121	9.60	2.16
DS	*P. purpureum (Pp)*	S4	Shoots	Stems	0.001	0	0.047	0.297	0.544	0.127	0	0.0057	9.25	
				Leaves	0.098	0	0.038	0.178	0.376	0.453	0	0.007		
		S11		Roots	0.015	0	0.043	0.371	0.757	0.563	0	0.072	3.60	18.49
		S11	Shoots	Stems	0.080	0	0.01	0.073	0.469	0.228	0	0.0018	3.77	
				Leaves	0.161	0	0.013	0.072	0.419	0.393	0	0.0014		

								BAF					MAI	CBCI
Season	Plant species	Sites	Plant tissues	Pb	Cd	Cr	Ni	Zn	Cu	As	Co			
RS	*C. benghalensis (Cb)*	S4		Roots	0.272	0	0.267	0.534	0.560	0.788	0	0.593	2.30	5.58
		S4	Shoots	Stems	0.055	0	0.018	0.006	0.485	0.578	0	0	1.77	
				Leaves	0.108	0	0.034	0.129	0.504	0.527	0	0.052		
		S9		Roots	0.280	0	0.182	0.655	0.867	1.587	0	1.140	2.08	3.02
		S9	Shoots	Stems	0	0	0.011	0	0.581	0.373	0	0	1.19	
				Leaves	0	0	0.028	0.06	0.503	0.507	0	0		
		S11		Roots	0.196	23.78	0.051	0.052	0.34	0.487	0	0.465	8.42	11.97
		S11	Shoots	Stems	0	24.29	0.003	0.004	0.078	0.201	0	0.0064	8.25	11.97
				Leaves	0	0.130	0.016	0.019	0.406	0.398	0	0.0065		11.97
DS	*C. benghalensis (Cb)*	S4		Roots	0.263	0	0.116	0.425	0.802	0.552	0	0.525	3.24	19.75
		S4	Shoots	Stems	0.06	31.154	0.21	0.016	0.522	0.294	0.141	0.009	3.06	
				Leaves	0.103	0	0.058	0.303	0.519	0.398	0	0.09		
		S9		Roots	0.141	0	0.147	0.755	1.547	0.537	0	1.094	3.03	1.25
		S9	Shoots	Stems	0.0028	8.991	0.015	0.019	0.402	0.123	0	0.003	2.15	
				Leaves	0.016	0	0.046	0.184	0.656	0.352	0	0.093		
		S11		Roots	0.333	0	0.012	0.076	1.623	0.728	0	0.039	2.49	1.52
		S11	Shoots	Stems	0.074	0	0.0006	0.002	1.619	0.305	0	0.002	2.31	
				Leaves	0.131	0	0.008	0.026	0.664	0.794	0	0.001		

Therefore, these observations were made only for site 11 during the rainy season, conversely, in the dry season where *C. benghalensis* exhibited the plants with the highest CBCI value. The lowest CBCI values observed in *C. benghalensis* at site 9 in the dry season (1.25) and in *P. purpureum* in the same site and in the same season (1.27) showed a high accumulation of individual heavy metal.

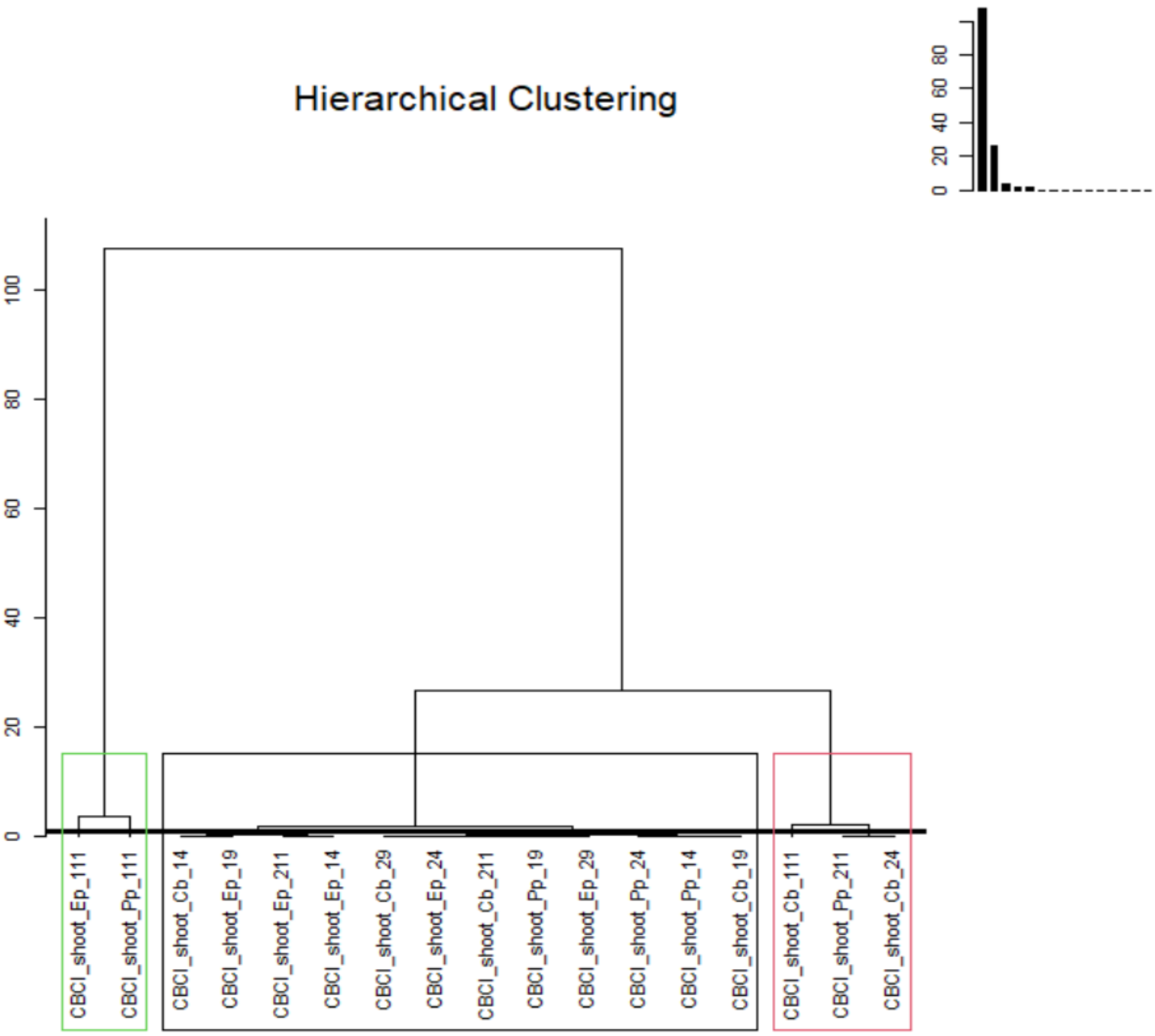

Fig. 50. Hierarchical cluster analysis (HCA) used CBCI for plant shoots at three sites during the rainy and dry seasons (1, 2). CBCI: Comprehensive Bioconcentration Factor Index, *Ep: E. pyramidalis*, *Pp: P. purpureum*, *Cb: C. benghalensis*, 4: site 4, 9: site 9, 11: site 11, 1: rainy season, 2: dry season.

Cumulatively, *E. pyramidalis* and *P. purpureum* appeared to be the best accumulators of multiple heavy metals during the rainy season, while *C. benghalensis* was the best during the dry season at the lowland sites. Metal bioaccumulation efficiency was presented following the decreasing order of: *P. purpureum* > *E. pyramidalis* > *C. benghalensis* (Fig. 51).

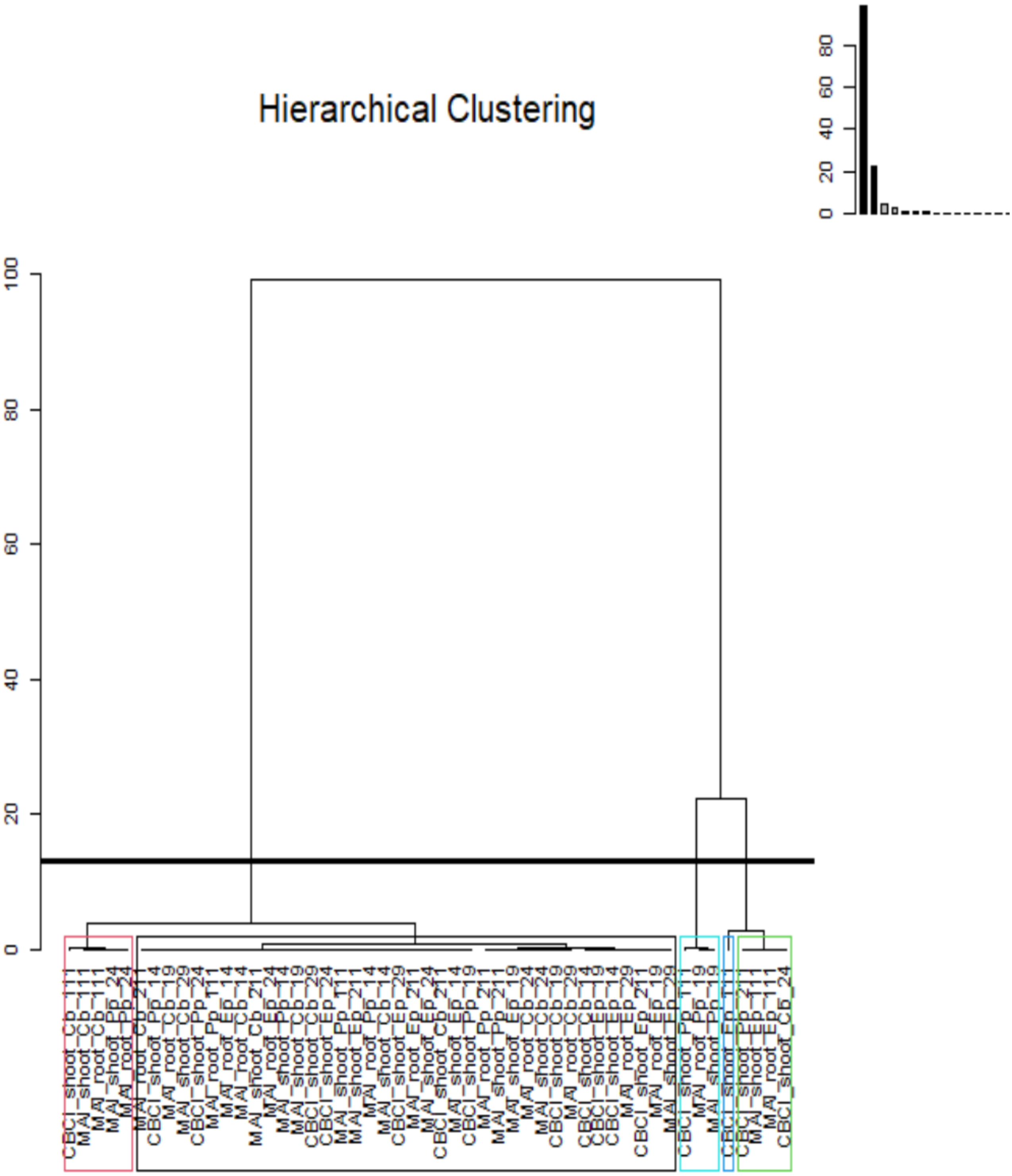

Fig. 51. Hierarchical cluster analysis (HCA) used CBCI and MAI of plant shoots and roots bioaccumulation efficiency at three sites during the rainy and dry seasons (1, 2). CBCI: Comprehensive Bioconcentration Factor Index, MAI: Metal Accumulators Index, *Ep: E. pyramidalis, Pp: P. purpureum, Cb: C. benghalensis*, 4: site 4, 9: site 9, 11: site 11, 1: rainy season, 2: dry season.

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III.1.3.3. Correlations between phytoremediation parameters and plant parts (PCA and Cluster Dendogram of plant parts and remediation parameters)

The dendrogram carried out with the results of the various parameters such as mobility ration (MR), translocation factor (TF) and bioaccumulation factor (BAF) used in this study showed that significant positive correlations existed between all metals analysed in plant parts and remediation factors of the three plant species studied. However, the shoots and roots of *C. benghalensis (Cb)* in sites 4 and 11 were closer together than those of the other two plant species. The principal component analysis (PCA) was applied and the analysis reduced the variables into three main principal components (PCs) explaining a total of 96.96% variability. PC1 and PC2 explained 62.88% and 34.08% variability respectively. Component 1 revealed a correlation between metals uptake and phytoremediation potential of heavy metals (bioaccumulation of roots, stems and leaves, as well as the translocation factor and the mobility ratio) along the sites of all three plant species and the three sites, except for the transfer factor of *E. pyramidalis* in site 11 (Fig. 52). Cluster diagram classified phytoremediation parameters in three groups revealing association between each of them and plant organs (Fig. 52).

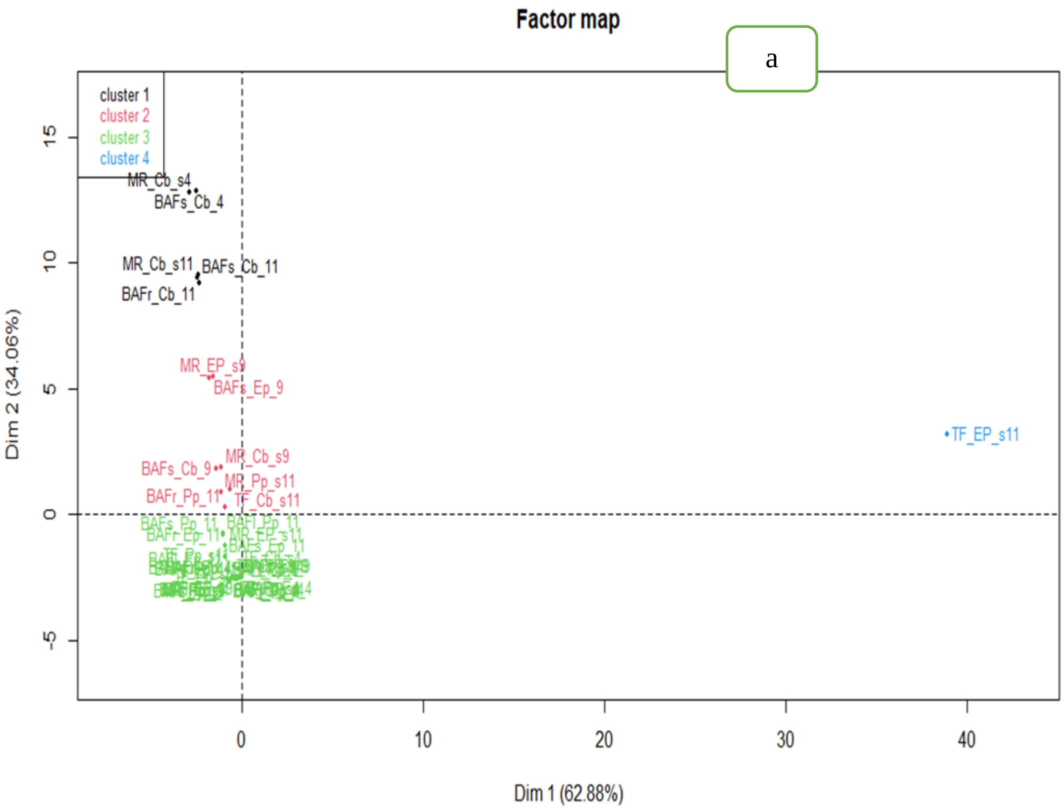

Fig. 52. PCA of plant parts and phytoremediation parameters of the study. (MR: mobility ration, TF: translocation factor, BAF: bioaccumulation factor, *Cb: C. benghalensis*, *Ep: E. pyramidalis*, *Pp: P. purpureum*).

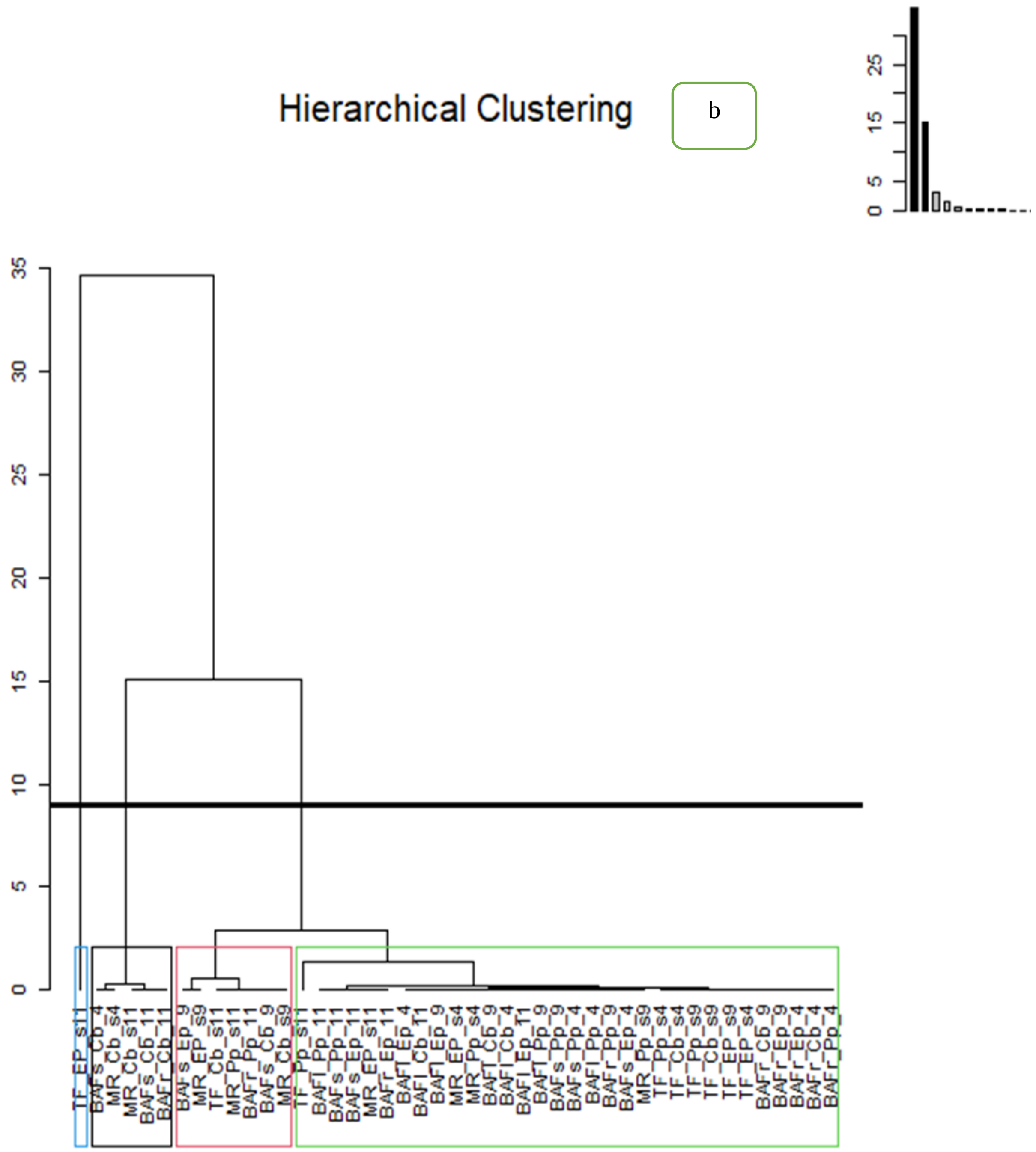

Fig. 53. Clustering of plant parts and phytoremediation parameters of the study. (MR: mobility ration, TF: translocation factor, BAF: bioaccumulation factor, *Cb: C. benghalensis*, *Ep: E. pyramidalis*, *Pp: P. purpureum*).

III.1.3.4. Correlations between plant species and the overall parameters of the study

In this study, the various soil, water and plant parameters such as the pollution index (PI), Integrated Nemerow pollution index (IPI), geo-accumulation index (Igeo), ecological risk index of soil (E^i_r), risk index of water (RI), ecological risk index of water (ERI), heavy metals evaluation index (HEI), mobility ratio (MR) and translocation factor (TF) were presented in the dendrogram. This dendogram was performed with the results of the plant species and was used to show the existed correlations between metal toxicity indices in soils and waters of all three sites and the plant species studied. The principal component analysis (PCA) showed that the plant species *E. pyramidalis (Ep)*, *P. purpureum (Pp)* and *C. benghalensis (Cb)* correlated with the soil and water toxicity indices such as RI, E^i_r, ERI in the different lowland sites (s4, s9 and s11). However, they showed no particular effect on the ecological risk of water (ERI), which meant that plants were much more involed with metals in lowland soils than with metals in lowland water (Fig. 54).

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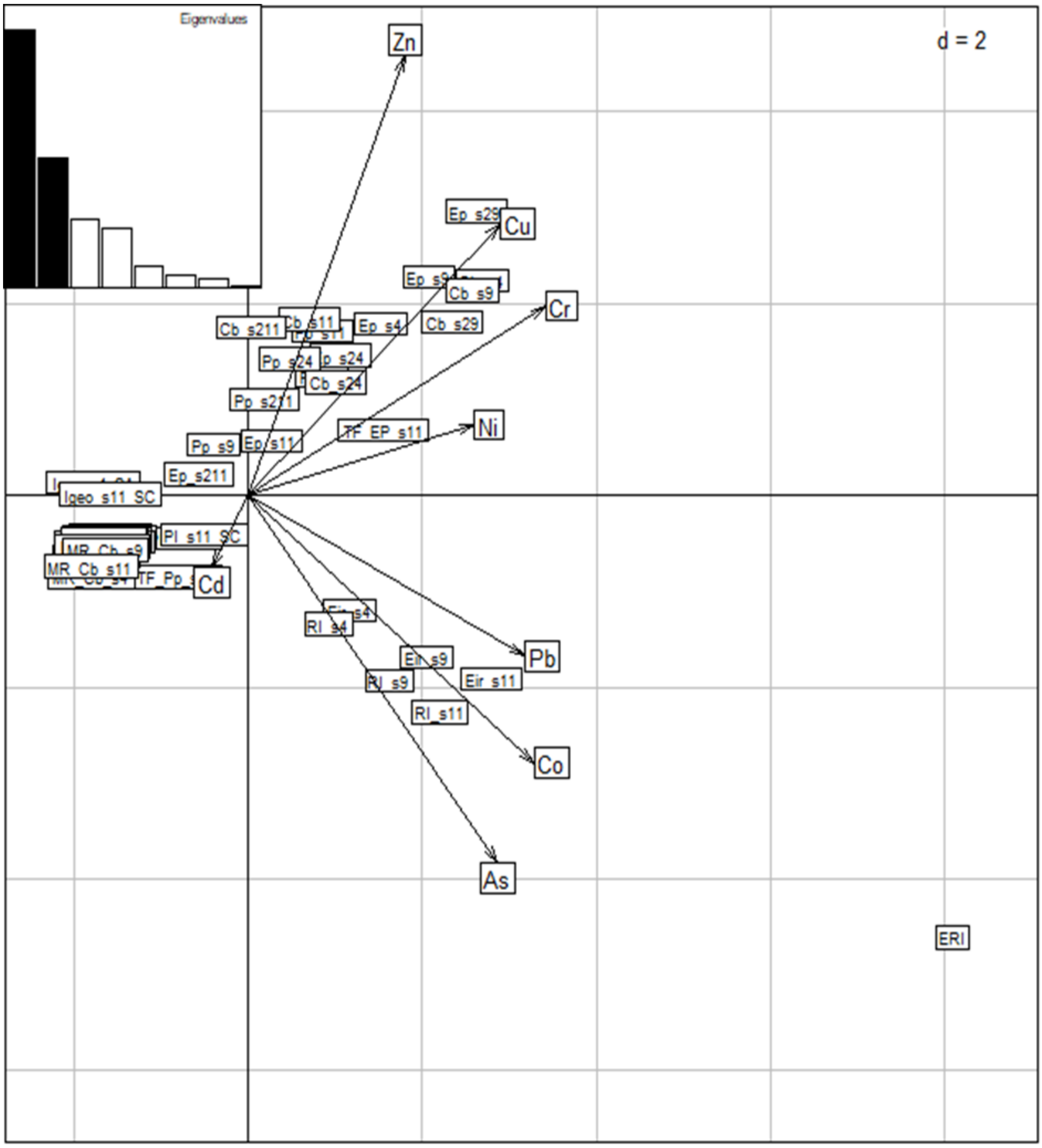

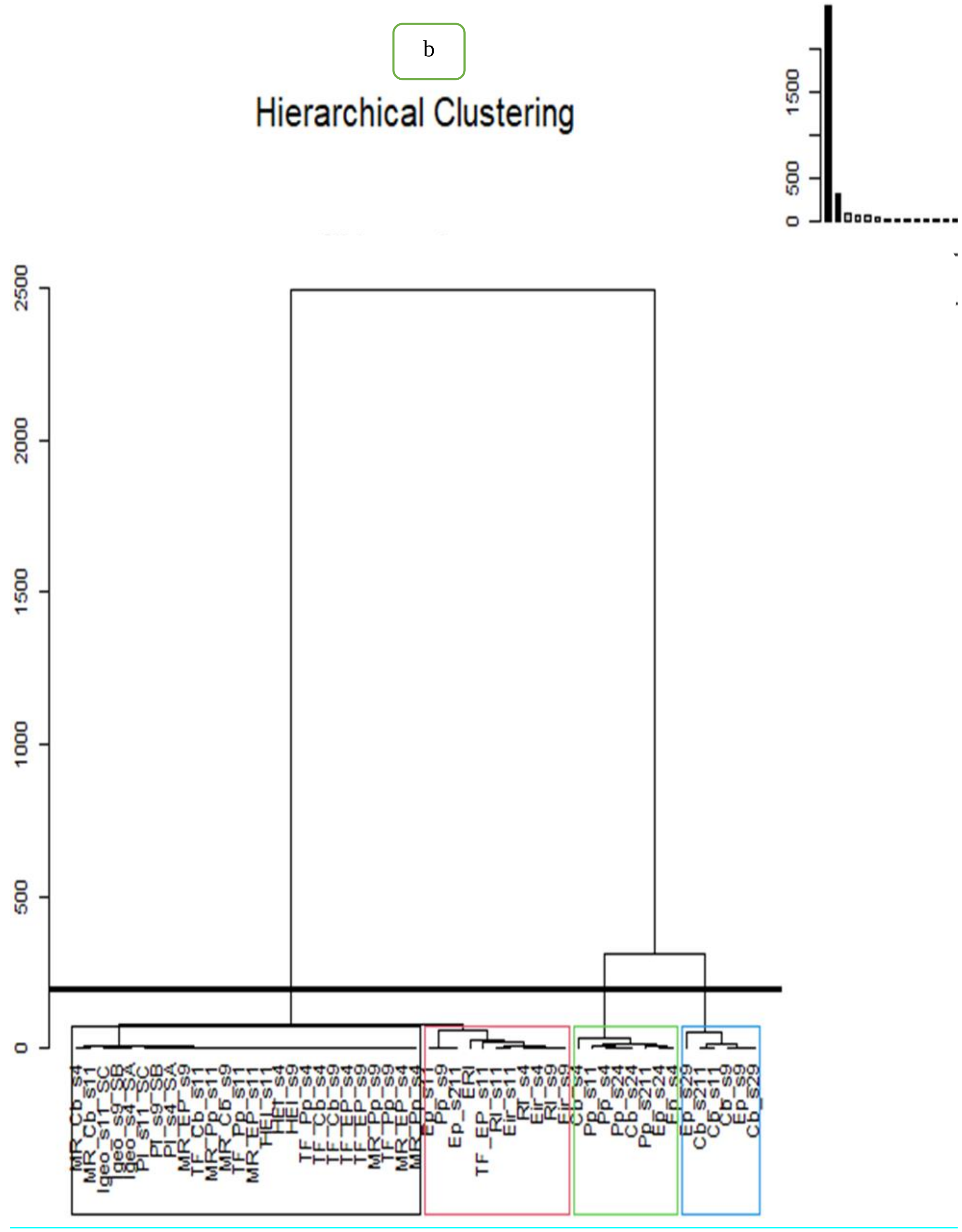

Fig. 54. PCA and clustering of plant species and the overall parameters of the study. (a. PCA, b. Cluster diagram. MR: mobility ration, TF: translocation factor, HEI: heavy metals evaluation index, PI: pollution index, IPI: Integrated Nemerow pollution index, Igeo: geo-accumulation index, E^{i}_{r}: ecological risk index of soil, RI: risk index of water, ERI: ecological risk index of water, *Cb: C. benghalensis, Ep: E. pyramidalis, Pp: P. purpureum*).

III.2. Discussion

III.2.1. Floristic diversity of Yaounde lowlands

III.2.1.1. Macrophytes' diversity of investigated sites during the rainy and dry seasons

In each of both seasons, the floristic affinity and similarity observed between the polluted sites could have resulted from the simultaneous presence of the ruderals on these sites (Messou et *al*., 2013). The macrophyte diversity of the contaminated sites compared to a control site, were shown to be significantly different (ANOVA test; $p<0.05$). This high macrophyte diversity of the lowland polluted sites could be attributed to the environmental stress due to the impacts of the anthropogenic activities such as dumping waste. The inflow of organic (degradable) and inorganic waste ending up in the lowlands adds nutrients which promote the growth of plant species. In response to the environmental factors as topography of the land, soil type, drainage characteristics, climate, and the level of contaminants present in the environment (Mijošek et *al*., 2020), some macrophytes growed in supposed metal-contaminated soils need to develop some degree of tolerance to metal toxicity in order to survive (Zhu et *al*., 2018b).

An assessment of the floristic inventory of the selected sites during the study revealed 189 plant species belonging to 138 genera distributed in 63 families during the rainy season and 139 plant species belonging to 103 genera dispatched to 39 families during the dry season. The main families found during both seasons in the study were Poaceae, Asteraceae, Fabaceae, Malvaceae and Solanaceae Cyperaceae and Convolvulaceae. This shows an increase in diversity compared with a previous study carried out in the Mfoudi watershed of Yaounde which recorded 135 plant species in 102 genera distributed in 44 families during both seasons, with a dominance of Asteraceae, Poaceae and Euphorbiaceae families (Tchinda et *al*., 2018). The increase in number of species and families can be attributed to the increase in human activities in and around the selected sites over the years, leading then to the decrease or local extinction of some species and the appearance of other most resistant ones.

The difference of plant families and taxa between the polluted sites and the control in each season (rainy and dry seasons) is likely due to the environmental conditions prevailing in each site during both seasons. The most represented families listed above in the lowland sites could be explained by the establishment of ruderal species that acclimate and adapt in these conditions of pollution. Njuguna et *al*. (2017) identified in Nairobi river 31 species with 11 species with the ability of metal bioaccumulators from 23 dominants families of Polygonaceae, Amaranthaceae, Commelinaceae and Cannaceae. Then, Messou et *al*. (2013) recorded as the most frequent families (36.9% of total taxa) of Poaceae, Euphorbiaceae and Cyperaceae.

Among these families, species of the family of Poaceae were most prevalent and reported as heavy metal tolerant (Tchinda et *al*., 2018).

The difference in number and abundance of species recorded during both seasons could be due to the difference in conditions prevailing in these areas. In the study carried out in 2018, most of the species were from the families of Asteraceae (16), Poaceae (12) and Euphorbiaceae (10), while for the present study, the main families were represented by Poaceae (23 and 19), Asteraceae (20 and 17), Fabaceae (14 and 8), Malvaceae and Solanaceae (12). This difference in term of species represented by families could be due to the modifications of the areas. The migration of people from one area to another especially the installation in lowland areas could have an impact on the stability of the diversity, leading to the destruction of the natural lowland ecosystems in order to build houses and shops. In addition, the increase in human activities in the environment could have raised pollution levels as most of waste was discharged directly into the river. Thus, the species found during this current study are adapted to and respond to this disturbance. These higher numbers of macrophyte families and taxa could also be due to these specific conditions prevailing in lowland areas. According to Shahabaldin Rezania et *al*. (2016), macrophytes adapted to the stressful conditions of heavy metal polluted soils since the vegetation of polluted and physically disturbed environments consist often of highly resilient species. The presence of the above families in the contaminated lowlands is the result of eutrophication due to the breeding activity, agriculture and pits latrine, which is going on in these areas. The families' taxa represented in the control indicate the forest recruits of the characteristic species of swampy lowlands in the tropical zone (Nusbaumer et *al*., 2018).

Site10 (Biyem-assi), site 8 (Ngousso) site 4 and 9 (Mokolo elobi and Mvan) were the sites where the highest number of species was recorded during the rainy season respectively 77±3, 70±4 and 55±3), while the lowest number was recorded in site 7 (Retenue pond (UYI)), site 2 (Nkolbisson-Nkol-nso'o) and site 3 (Nkolbisson-IRAD) with values of 22±2, 29±2 and 33±4 bits/ind respectively. However, during the dry season, the highest number of species recorded was in site 8 (Ngousso), site 3 (Nkolbisson-IRAD) and site 7 (Retenue pond (UYI)) with 50±2, 44±3, and 39±3 respectively, while the lowest number of species was recorded in site 5 (Messa) with 13±1, site 11 (Atemengue pond) 22±2 and site 1 (Cite-verte) 27±3 bits/ind. The variation in the number of species per site could be explained, by the size of the different sites studied, but also by the similarities or differences in the prevailing conditions in each of these areas, which could favour one specie over another. These values suggest that these different lowland sites are rich in specie diversity. According to Gaines et *al*. (1999); Fayiah et *al*. (2020), a river or site can be regarded as poor specie-rich or high specie-rich when the values range from 1.5 (for low species richness) to 3.5 (high species richness), although in occasional cases, these values could exceed 4.5. The elevated specie diversity within this ecosystem during both seasons could be explained by the nature of the zone, which is an open area with photosynthetic vegetation important for wildlife use, and are therefore important sites for biodiversity conservation. These lowland ecosystems acted as biological filters, retarding pollutants from entering lakes, rivers and groundwater (Qu et *al*., 2019; Nguyen et *al*., 2019). Another explanation for the elevation in terms of species between the sites studied could be the impacts of activities surrounded these areas (especially in sites 5, 1 and 10 with the constructed wetlands treatment plants rehabilitate in Yaounde and sites 3 and 4 with the development of agriculture) or water fluctuation and constant floods during both seasons, although the fluctuation of water brings new types of wastes or not in lowland areas. Considered as a natural form of disturbance in lowlands, water level fluctuation has several quantifiable components, including intensity and frequency of flood (Wang et *al*., 2015). According to Schneider et *al*. (2018), water movements, caused by waves or currents, influence macrophytes in a complex way, as they affect plants both directly and indirectly. Compared to the control, the polluted sites could be an indicator of the state of pollution in Yaounde lowlands and might suggest a process of macrophytes invasion of the eutrophic inland valleys in the urban zone.

The value of Simpson’s index of diversity during the rainy season showed that sites 10 and 8 (Biyem-assi and Ngousso) were more diverse than the other sites, while in the dry season there were site 8 and 7 (Ngousso and Retenue pond), which means that the individuals in the community are distributed more equitably among these species (Ma & Ellison, 2018). Indeed, the greater the value, the greater the sample diversity as the Simpson's diversity index is a measure of diversity which takes into account both richness and evenness.

Sorensen’s index value obtained when comparing all the polluted sites and control during both seasons was much lower than 50% (K_{RS}<K_{DS}<50%) indicating the non-influence of the season on the specie diversity in term of similarity between the sites. Therefore, it was higher in the dry season (K=12.24%) than the rainy season (K=7.9%) between sites showing the less similarity between sites during the rainy season than the dry season. This results confirm the observation of Ngueguim et *al*. (2010), which presented that the smaller sorensen similarity index is, the less similar the site is. Sorensen’s index value (K_{RS}<K_{DS}<50%) indicates that plant communities are different and there is no similarity in the flora between the two types of sites investigated. This trend was also observed between each polluted site and the control except in site 3 (Nkolbisson lake) in the rainy season and site 5 (Messa) in the dry season. This can be the results of the local environmental conditions (temperature, drainage, nature of soils grazing, cutting and compaction) which could be the main factors explaining the difference or similarity in plant communities between polluted and control sites (Zhang et *al*., 2017).

Amongst all sites studied, the most abundant species found were *E. pyramidalis, I. aquatica, C. benghalensis* and *P. purpureum* during the rainy season, while there were *E. pyramidalis, C. benghalensis, P. purpureum, P. maximum* and *I. aquatica* during the dry season in site 4 (Mokolo-elobi), site 11 (Mvan) and site 9 (Atemengue pond Obili) during both seasons. At theses sites which were found the species with a high level of abundance-dominance, three were common to those found by Tchinda et *al*. (2018) in their study on the influence of the origin of water pollution on the floristic diversity of macrophytes of the Mfoundi lowlands inYaounde that were *P. purpureum* (14.9%), *C. benghalensis* (9.3%) and *E. pyramidalis* (8.4%) which had as well a high level of abundance-dominance.

In general, we observed that the most abundant macrophytes were not the most frequent on polluted sites also in the control site and vice versa and this for both seasons. These results could be used to explain the differences in specie diversity observed between polluted and control sites. Species diversity is an important index correlating the number of species and their distribution. Several authors said that the indicative value of a specie depends on its overall relative frequency and abundance in sampling (Messou et *al*., 2013). However, abundant species can as well be characterized as tolerant to heavy metals in the tropical zone (Tchinda et *al*., 2018; Nguemte et *al*., 2018). Abundance being the best measure of the degree of tolerance for species growing in disturbed habitats. Thus, species that are present and dominant in such areas should be those that have an adaptation strategy. According to Eid et *al*. (2020), heavy metal tolerance in plants is made possible through metal sequestration. This by production of organic compounds, compartmentalization in cell compartments, metal ion efflux and organic ligand exudation. Such plants are referred to as “polluo-tolerant plant species” (Schwoertzig et *al*., 2016; Bonanno et *al*., 2017).

III.2.1.2. Potentially useful plants for phytoremediation of heavy metals

In this study, the potential useful macrophytes species for phytoremediation of heavy metal contaminated sites included species from the families of Poaceae, Commelinaceae, Nympheaceae, Euphorbiaceae, Polygonaceae and Onagraceae. Among species of these families, some have showned a potentiality to decontaminate heavy metal polluted soils and water. Indeed, *E. pyramidalis* (Poaceae) and *L. stonifera* (Poaceae) applied together achieved 61.52% and 93.22% higher removal for Cd and Pb from wastewater (Syokor et *al*., 2016).

According to Adesuyi et *al.* (2018), *I. aquatica* var edulis (Convolvulaceae) can concentrate higher level of lead while *C. benghalensis* (L.) (Commelinaceae) sequester high levels of chromium. *I. aquatica* (L.) (Convolvulaceae) was identified to be suitable for phytoextraction of Copper and Manganese in soils contaminated by heavy metals (Mohotti et *al*., 2016). *C. benghalensis (*L.) (Commelinaceae) and *P. maximum* Jacq. (Poaceae) possess 50% of capabilities for tolerance to heavy metals particularly Cd (Sekabira et *al*., 2011) in contaminated soil sites. *P. purpureum* (L.) (Poaceae) indicated heavy metal accumulation in this order Cr>Zn>Cu>Pb at different parts of his tissues (roots, shoots, leaf) on tannery soil contaminated (Juel et *al*., 2018). The ability of plants to tolerate and accumulate these metals may provide the bases for their phytoremediation usefulness.

The major macrophyte species encountered in lowland polluted sites belong to the families of Poaceae, Amaranthaceae, Asteraceae, Convolvulaceae, Euphorbiaceae, Cucurbitaceae, Cyperaceae, Musaceae, Araceae, Acanthaceae, Polygonaceae. According to Dhaliwal et *al*. (2020), the dominant hyperaccumulator families included Asteraceae, Euphorbiaceae, Fabaceae, Flacourtaceae, Brassicaceae, Caryophylaceae, Lamiaceae, Poaceae, and Violaceae. In the lowland polluted sites, four of these dominant families were recorded. According to the literature, most plant species grown on these sites showed a capacity to tolerate heavy metal accumulation (Ghazaryan et *al*., 2019). The species of these families can be grasses and herbs that according to the literature are highly efficient in phytoremediation because they possess a fibrous, extensive and diversified root system, which could give them the advantage of being more competitive during the colonization of polluted media (Prokop'ev et *al*., 2014; Dinu et *al*., 2020).

III.2.2. Level of heavy metal contamination in soil, water and plants of lowland in Mokolo-elobi (site 4), Mvan (site 9) and Atemengue pond Obili (site 11) of Yaounde during the rainy and dry seasons

III.2.2.1. Soil physico-chemicals and heavy metal concentrations in the three lowland sites

Soil pH determined the acidity or alkalinity (basicity) of the soil solution and played an important role in the availability of nutrients to the plants. Many plant species adapted to a range of soil pH generally from 5.5 to 7.5. The study of the pH (KCl) and pH (H_2O) of the lowland soils of different sites (4, 9 and 11) showed a significant difference ($p<0.05$) during both seasons. The variation of ΔpH (pH (KCl) – pH (H_2O)) was negative throughout the study. This indicated that the net charge on the exchange complex was negative, and thus exhibit cation exchange capacity. However, according to Yerima & Van Ranst (2005a) some tropical soils due to intensive rainfall and weathering are dominated by positive charges with anion exchange capacity predominant. During the dry season, the soil samples were slightly acidic to neutral than in the rainy season where the pH (H_2O) was basic (min. 6.75; max. 7.64). This showed an increase in pHs values (pH (KCl) and pH (H_2O)) compared to a previous study carried out in 2015 in the Ntem watershed of Yaounde which obtained in the surface soil pH (H_2O) from 4.11 to 5.96 and pH (KCl) from 3.58 to 5.36 (Defo et *al*., 2015). The increases of the pH values in the three sites studied in this lowland ecosytem with time can be attributed to the geomorphology of each site, type of activities developped around the sites, type of raw wastewaster discharged on the sites and invasive plant species present. The tendancy of soil to be acidic was probably due to the degradation of organic matter and the subsequent formation of carbonic acid. Therefore, the acidity of soil can be related to the leaching of the basic cations to the deeper layers. Another explanation could be the nature of the lowlands, which were sites with high water flow (especially during the rainy season), and where the soils are generally older and well developed on stable geomorphology (flat areas or gentle slopes) (Brosens et *al*., 2020). According to Neina (2019), higher soil acidity favoured the presence of cations in the soil. Among soil properties, soil pH had strong effects on the solubility and speciation of metals in the soil and especially in the soil solution (Fanrong et *al*., 2011), while organic matter content had a strong influence on cation exchange capacity, buffer capacity as well as on the retention of heavy metals.

In all the sites studied, EC and TDS were significantly different between the three sites studied. This implicated that as the lowlands studied received various types of water from different sources, mineralization process is not important due to movement of water and the leaching of soils. As EC is used to estimate the concentration of soluble salts in soil, the study carried out by Salem et *al*. (2020) on the physico-chimicals properties of soils and HM concentrations during the wet and dry seasons in Lybia showed also a lower value of EC due to low salt content which was washed down during the irrigation processes. Thus, compared to WHO standard, the lower values of EC and TDS in lowland soils might be explained by the dissolution and the precipitation of salts complex and their transport to plants. Because of this, the variability of soils solution in terms of electrical conductivity also affected the salinity, reason why the salinity effect of all lowland soils was almost negligible.

TOC is one of the factors that affects and regulates the presence and behaviour of other chemical components, such as metals in sediments (Nguyen et *al*., 2019). Consequently, there will be a tendency of higher metal accumulation at sites with fine-grained sediments and increased OC concentration. This study showed that in site 4, during the rainy season, the values of OC (1.67%), OM (1.42%) and CEC (7.07) were higher than these values in the same site during the dry season 1.03%, 2.05% and 4.26% respectively, while the opposite was seen in the other sites (9 and 11) during both seasons. This showed the influence of both seasons on the lowland soil composition. The results obtained in this study showed a decrease in TOM and CEC values compared with a previous study carried out by Defo et *al*. (2015) in the Ntem watershed of Yaounde which recorded in the surface soil the values of TOM from 6.90 to 9.80% and CEC from 10.70 to 14.70 meq/100g. The decreasing value of TOM and CEC can be attributed to the modification of soils with time and in lowland soil organic matter breaks down very slowly because microorganisms necessary for decomposition cannot mineralized where there is lower or no oxygen. In addition, as CEC is influenced by organic matter and clay content, while soils with low CEC have low ability to hold water (sandy soils).

The highest percentage value of organic carbon was observed in soil C collected from site 11 and lowest value observed in soil A from site 4 in the dry season. High organic carbon content indicated that metals were more likely to form metal chelate complexes in organic matter, which would also lead to fewer metals being available for plants (Wiatrowska & Komisarek, 2019). Mayuri et *al*. (2016) have reported that some fractions of metals can be bound with organic content therefore, the determination of organic carbon in soil is very important. Soil organic matter can influence the uptake of heavy metals by plants. Site 11 in the dry season presented the highest organic matter. This can be attributed to fertilizers especially organic fertilizers (Azzi et *al*., 2019). The soil clay contents were less than 20% in sites 4 and 9 during the dry and rainy seasons and such soils are classified as coarse textured soils (Akortia et *al*., 2019). According to Zhang et *al*. (2014) silt and clay fractions have particularly important influence on the transport and storage of heavy metals within fluvial sediments. In addition, the OM had a weak positive correlation ($r = 0.411$; $p>0.05$) with the clay fraction of soil. This was an indication that the distribution of organic matter in the soils was not influenced by clay. This variation might be attributed to the constant addition of organic matter from varying anthropogenic activities such as the application of poultry manure, chemical fertilizers, and wastewaters. Fomenky et *al*. (2017) on their study on physico-chemical properties of soils and some water sources on the Eastern Flank of Mount Cameroon, observed a weak positive correlation ($r= 0.218$; $p>0.05$) with the clay fraction of soil. However, organic matter contents can be incorporated into the interface of the clay minerals. According to Aranda et *al*. (2011), sediments were considered to be rich in organic matter when the content of organic matter was higher than 2%. Indeed, the higher values of organic matter observed, was then contributed to the pollution of the lowland areas.

Particle size defined important factors that could affect TOC content in soil (Akortia et *al*., 2019). Since smaller particles have larger surface area to volume ratio, they have a larger binding capacity for the adsorption of organic carbon. Furthermore, these organic matter coatings are common in fine sediments, and they bind a variety of trace elements (Mijošek et *al*., 2020). In this research, the highest TOC content was observed in site 4 during the rainy season, where the smallest grain size and the average TOC (1.67%) was obtained, compare to other sites. On the contrary, TOC was higher during the dry season in site 11 (3.58%) than the others. The elevation of TOC in the dry season could be due to the infiltration of wastewater in soil and the deposit of organic matter on the sediment surface. Another explanation could be the effect of water stability during this season where the water flow was slow and all deposits have enough time to sediment and accumulate on the sediments.

In these studied areas, the mean concentration of Pb, Cd, Zn, Cu and As determined were within WHO, EU standard and MINEPDED standards in soils. The statistical analysis showed no significant difference ($p>0.05$) on the concentrations of the studied metals between the rainy and dry seasons. However, the average concentrations of Cr (285.13±62.7) and Co (20.23±1.71) in all soils were higher than the threshold values. The mean concentration of Ni (80.29±24.88) was above the standard value only in soil of site 11. These indicated that, the studied soils were contaminated with Cr, Ni and Co that might cause a toxicity of the lowland environment. In a study carried out by Defo et *al*. (2015) in a Ntem watershed of Yaounde, similar observations were for Pb (162.8±12.67), Cr (77.28±9.58) and Ni (84.37±8.09) where the mean concentrations were higher than the threshold levels allowed for soils. Compared to this study, the mean concentration of Pb was above the standard value and the mean concentration of Ni was higher than the norm values only at the station (P3) or site. These high values of metals enlightened the fact that nothing has changed over time, but that the level of pollution increases continuously. Another study carried out by Mandeng et *al*. (2019) in Kienké and Tchangué watershed in the sourthern region of Cameroon showed that the average value of Ni 72.4 mg/kg and 217.28 mg/kg was above the EU standards value (≈ 68 mg/kg) and this was attributed to the natural weathering and leaching of rocks. On the other hand, in the soil sediments of the lowland of Ilova village (Croatia), Mijošek et *al*. (2020) found high concentration of Cr above the Canadian Council of Ministers of the Environment value (SQG= 37.3 and PEL= 90 mg/kg) and explained it by the presence of fish ponds and farms. It assumed that according to Sarker et *al*. (2016), poultry and tannery wastes could be added as supplements in the fish feed, whereas $K_2Cr_2O_7$ has an important role in cleaning.

Chromium concentration in all lowland soils was higher than other metals. In each site studied, Cr was higher than the WHO limits (100 µg/g) for soils. This can be a consequence of a direct discharging of untreated wastes and wastewaters coming from agro-industries (SOFAVIN), lixiviats from waste bin, fertilizers and chemical products used in the laboratories. However, high level of Cr in site 11(C) (248.38 and 236.72 µg/g) during the rainy and the dry seasons respectively indicated its higher input in soils. These might be originated from chemical products used in the CHU mortuary for cleaning corps, reagents used in the laboratories for diverse analysis and fertilizers used for agriculture and tests experiments by students of the University of Yaounde I. According to Islam et *al*. (2015a), Ali et *al*. (2018), in their studies on the assessment of HM contamination in surface sediment and toxic metals in water and sediment of Pasur River, both in Bangladesh, reported that the higher level of Cr observed in sediments was a consequence of direct discharging untreated wastes from petroleum, fertilizers and textile industries. The occurrence of Cr in soils could be due to waste disposal consisting of lead chromium batteries, coloured polythene bags, discarded plastic materials and empty paint containers (Amos-Tautua et *al*., 2014). This indicated that the higher level of Cr in soil might originate from urban and industrial wastes. The toxicity of Cr has negative impacts on the growth of plants by impairing their essential metabolic processes such as chlorophyll biosynthesis, photosynthesis and plant defensive system (Sharma et *al*., 2020).

Concerning nickel, the highest concentration was observed in soil of site 11 (Atemengue pond Obili) (80.29 µg/g) during the rainy season and it was higher than the target values (50 µg/g). Target values were specified to indicate the environmental quality of soil with the assumption that there was a negligible risk on the ecosystem (Parihar et *al*., 2020). Nickel was a widespread metal/metalloid in the environment. Its sources can be electroplating, non-ferrous metal, paints and porcelain enamelling. The effects of Ni in the human organism are cardiovascular diseases, chest pain, dermatitis, dizziness, dry cough and shortness of breath, headache, kidney diseases, lung and nasal cancer and nausea (Fashola et *al*., 2016). High Ni concentration might be attributed to its accumulation at the surface of sediments from deposition of chemical compound from the laboratories and hospital located upstream of the lowland. Adiloğlu et *al*., (2014) attributed the Ni accumulation on the surface of sediment to artisanal gold mining deposit, fuel consumption in the motor vehicles and agricultural activities. Therefore, a large part of the nickel in soil formed from nickel-rich rocks belongs to the silicate and it is not an available form. The study of Mandeng et *al*. (2019) assumed that, in Cameroon context Ni accumulation could also be associated with the alteration of ultramafic rocks such as peridotite. Nickel can cause cardiovascular disease, dermatitis, kidney disease, lung fibrosis and respiratory cancer, in the human body (Hasnine et *al*., 2017).

The correlation between HMs and physico-chemical properties of soil showed that there was a positive significant difference at ($P<0.01$) for pH and Cu (r=0.591**), T°, EC and TDS with As (r=0.773**, r=0.891** and r=0.911**) and Cd (r=0.756**, r=0.895** and r=0.881**). CEC, OC, OM and silt were correlated with Cr, As, Ni, Cd, Co and Zn, while sand was positively correlated with Pb (r=0.592) and Cu (r=0.671**), and clay only with Co (r=0.828**). The significant positive correlation between the metal concentrations and the physico-chemical properties of soil may be due to the variation of temperature in different seasons, which affected indirectly the absorption of heavy metals in the soils. A study carried out by Salem et *al*. (2020) on the assessment of physico-chemical properties and concentration of heavy metals in agricultural soils fertilized with chemical fertilizers in southern west of Libya found a significant correlation between pH, EC, CEC, OM and HM contents of soil. However, they explained that positive significant correlation found CEC-OM (r=0.960), CEC-Fe (r= 0.924), OM-Mn (r= 0.931), OM-Fe (r= 0.966) and Mn-Fe (r= 0.987) were due to the high affinity of metals to soil organic matter. Nguyen et *al*. (2019) showed that pH was significantly and positively correlated with B and Sr, while negatively correlated with Al, Cd, Co, Fe, Ni, Pb, and Zn and also reported that the increased concentration of heavy metals such as Al, Fe, Mn, and Zn could be observed when pH was below 7 (Brady & Weil, 2017). The negative significant correlation was found between pH (H_2O) - As (r=-0.609**), sand and Cr, Ni, Co (r=-0.611**, r=-0.688**, r=-0.915**), and clay and Pb, Cu (r=-0.643**, r=-0.678**). The negative correlation might be attributed to the low content of organic matter, which plays a significant role in determining the availability and mobility of HMs in soil, and/or to the nature of soil texture.

Similarly, significant positive correlation between the HMs themselves suggested the similar and identical origin of the HMs, and indicated that the anthropogenic activities could enhance the mobility of these elements (Salem et *al*., 2020). Soil parameters had common sources or mutual dependences, subject to certain factor controls.

The geo-accumulation index was used to determine anthropogenic contamination in river sediments (Gupta et *al*., 2014; Ji et *al*., 2018). The calculated Igeo values showed for the majority of metals (Pb, Cd, Ni, Zn, As and Co) indices below zero, indicating that the concerned metals did not contaminate the soils. Salem et *al*. (2020) explained that due to the absence of pollutant sources such as anthropogenic activities, heavy traffic or irrigation with polluted waters, the low values of Igeo might have also been raised from the metals' geochemical and biological interactions and variations. However, all three sites were heavily contaminated by Cr (1.61<Igeo<2.89) and moderately contaminated by Cu (0<Igeo<1.16). At all three sites, the ranges of Igeo values for the metals were relatively wide, confirming the variability of the lowland soil properties and sources of heavy metal pollution. In a study carried out by Defo et *al*. (2015) in a Ntem watershed of Yaounde, the results showed that Igeo values for Pb, Cr and Ni were higher than 0 and less than 1 (0<Igeo<1). The Igeo values for Pb and Cr were ranged between 0.13 and 0.19, while it was about 0.1 for Ni. The authors explained that these values indicated that the given urban soils were contaminated by metals (Pb, Cr and Ni) derived from anthropogenic sources.

The increase of Cr and Cu concentrations in these soils could only be associated to the geochemical weathering (natural source). The Igeo values for Cr and Cu were ranged between 0.49 and 2.88. This could indicate the contamination of the lowland soils by metals (Cr and Cu) derived from anthropogenic sources. The high levels of Cr possibly originating from the smelting of ferrochrome and wastewater (Addo-bediako, 2020). Wei et *al*. (2019) in their study on the concentration and pollution assessment of heavy metals within surface sediments of the Raohe Basin in China, found that sites where there were mining activities produced a large amount of wastewater, had the highest values of Igeo for Pb and Cu, which fell into class 2 and 6, respectively. They also observed that a mean Igeo value of Cu (2.30) showing moderate to strong contamination. The finding implies that there was high level of soil heavy metal contaminants in mine sites, thus high health risk (Obasi et *al*., 2017; Xiao et *al*., 2017).

The risk analysis of geo-accumulation index (Igeo) indicated that the heavy metals Cr and Cu were the main pollution factors and each element of the pollution degree followed the order of: Cr>Cu>Co>Zn>Pb>Ni>As>Cd. Defo et *al*. (2015) in an urban watershed of Yaounde Cameroon observed in descending order the contamination of soils as follows: Pb>Cr>Ni>Cd, while Wei et *al*. (2019) (sediment of Raohe Basin China) determined the pollution degree followed the order of: Cu>As>Pb>Cd>Cr>Zn, Proshad et *al*. (2019) found the Igeo of Cd>Cu>Ni>Pb>As>Cr in agricultural soils of Bangladesh and Proshad et *al*. (2018) (Korotoa river in Bangladesh) indicated the decreasing order of Igeo values of As>Cu>Ni>Cr>Pb>Cd. The variation in terms of Igeo of different sites around the world depends of the presence or not of the specifical pollution source, which affected differently the environment of each of these sites. As the region is facing the increase of urbanization, industrialization and manufacture, the increase of contaminants are of greatest concern. According to Hanfi et *al*. (2020) the contamination with heavy metals depends on anthropogenic sources of the land-use, the rapid urbanization. Moreover, the significant factors are traffic and population volume.

In this study, the results showed that the studied lowland soils were not polluted by Cd and As (PI<1). The presence of these metals in soils could be associated with geological processes. According to Hu et *al*. (2020), geological processes provide additional links to ecosystem properties, which are important but show divergent effects on biodiversity and ecosystem functions. For example, bedrock and weathering exert considerable direct effects on biodiversity, while they indirectly influence ecosystem functions via interactions with biodiversity and contemporary environments. However, the highest PI values were observed with Cr (11.18), Cu (3.36) and Co (2.73), higher than 1 (PI>1) in all three sites, indicating the similarity of activities causing pollution in lowland soils. All these sites are located around metal workshops, iron and metallic recycling shops, agricultural farms, waste dumps, industries, laboratories and mechanical garages. Defo et *al*. (2015) observed that soils were polluted with Pb, Cr and Ni in the sampling stations with the PI values of 6.48, 2.99 and 2.29 respectively, where (PI>1), and not polluted with Cd (PI=0.11) in the Ntem catchment area in Yaounde. These authors explained that old batteries, paints, pigments and electroplating, plastics, pecticides and fertilizers from urban waste contain high levels of heavy metals such as Cr, Pb, Ni and Cd. The increase in chromium level over time in the lowlands in the present study could be explained by the increase in the use of this element in the manufacture of various products used such as batteries, reagents, agricultural fertilizers in cities. As these products were not well managed, they were found everywhere in the environment and in landfills. Furthermore, in large cities of developing countries such as Yaounde, recycling techniques for chrome products were not well developed and applied and consequently all types of waste were found in the lowlands. Müller et *al*. (2020) explained that anthropogenic activities associated with urban development generated waste and pollutants on catchment areas that can be washed into water bodies during rainfall events. Furthermore, Fosu-Mensah et *al*. (2018) in the study of the assessment of contamination and distribution of heavy metals in surface soils and plants along the west coast of Ghana found that the highest contaminated soils were by Se, As and Pb. The pollution index values showed PI - Se (4.44), PI - As (2.46), PI - Pb (2.26) for contaminated soils and PI - Cu (0.24), PI - Hg (0.017) and PI - Zn (0.82) for uncontaminated soils, which were attributed to the impacts of crude oil drilling activity on the environment. The differences observed between the observed metal pollution and the selected metals in each of these studies as well as the level of contamination of the soils by a specific metal could reflect a direct and specific link between the metal and the composition of the environment in different sites.

The Nemerow Integrated Pollution Index (IPI) was greater than 3 (IPI>3) on all sites indicating a very high metal contamination of the soils. The IPI values showed 8.06 for soil C, 5.79 for soil B and 3.41 for soil A, indicating the level of metal pollution in sites 11, 9 and 4 respectively. This result shows a similar pattern to that obtained by Defo et *al*. (2015) who found that the IPI of different soil samples in the Yaound**e**-Cameroon Ntem catchment area was above 3 (between 1.68 and 4.94). The increased pollution level in site 11 (Atemengue Obili pond) could be due to chemical pollution and extensive eutrophication in this area. Another explanation for the increase in IPI levels in the lowland sites studied could be attributed to soil properties. Indeed, lowland soils belong to the tropical soils whose weathering processes affect the vulnerability of soils to pollution. According to Yerima et *al*. (2014), weathering processes generate acidic soils with low buffering capacity and favour the retention of heavy metals in their exchange complex. In addition, the study of Kang et *al*. (2020) on the pollution characteristics and ecological risk assessment of heavy metals in paddy fields of Fujian Province, China, showed that the IPI was subject to a low pollution level (1<IPI<2), but the IPI level was in a warning line area of (0.7<IPI<1) in Yongcun and Zhaoan County. Compared with the current study, the authors attributed the low pollution level to the long-term industrialization and development of green economy in the region, such as the construction of an ecological province.

The single potential ecological risk index (E^i_r) values for each metal were below 40, indicating a slight risk of metals in lowland areas. Cr was the metal with a considerable potential risk to lowland soils (22.36), followed by Cu (16.8). Despite the fact that the toxicity response coefficient of Cr is smaller compared to that of Cd, which has the highest toxicity coefficient value, the increased risk level of Cr in lowland soils would indicate the source of this element. Indeed, chromium pollution can originate from various sources, the main ones of which can be attributed to the waste load in the environment such as sewage sludge, agricultural, industrial and hospital wastewater. The study by Mandeng et *al*. (2019) in the Kienké and Tchangoué watersheds in the southern region of Cameroon showed a low potential ecological risk factor ($E^i_r < 40$) for Cd (3.83 and 2.964), Pb (29.28 and 29.88) and Hg (4 and 2.72), while Zn (46.66) and Cu (50.42) showed a moderate and significant potential ecological risk. The authors indicated that nickel had the highest (E^i_r) level in both catchments. The (E^i_r) ranking followed the descending order of Cr>Cu>Co>Pb>Ni>As>Zn>Cd in the lowland soils. Kolawole et *al*. (2018) in their study on a heavy metal contamination and ecological risk assessment in soils and sediments of an industrial area in southwestern Nigeria, showed a high potential ecological risk for Cd, Pb and Cu, following the order of Cd>Pb>Cu>Ni>Cr>Mn in the industrial area.

They found that cadmium was the main influencing factor causing the potential ecological risk. All these studies showed the similarity and difference in the presence of trace elements in the environment. Indeed, the present study found similar observations. Furthermore, the contribution of a single metallic element would result in an ecological risk to the environment even if it were of small magnitude. This puts the emphasis on the impacts of single metal on the environment.

The potential ecological risk index (RI) for all metals (Pb, Cd, Cr, Ni, Zn, Cu, As and Co) at three sites was 46.64, 54 and 65.15 below 150, indicating a slight level risk of metals in lowland soils. The RI described both the ecological risk caused by a single pollutant and the overall risk or contamination by different types of pollutants. Compared to the pollution index, the metals studied did not much affect the ecological environment of the Yaounde lowlands. In a Fujian province of China, Kang et *al*. (2020) observed the RI value of 49.06, which showed a low risk on the environment. Mandeng et *al*. (2019) observed in sediments RI values of 378.83 and 1311.38 in the Kienke and Tchangoué catchments respectively, indicating significant and high ecological risk of metals in this region.

Principal component analysis (PCA) enhanced subtle but significant single element anomalities and helped to discriminate between element associations, those with different underlying factors controlling their distribution. The results of the principal component analysis (PCA) showed that there could be two main sources of metal enrichment in Yaounde lowland soils during the rainy and dry seasons. The presence of Ni, Cr, Co, Zn, As, and Cd from PC1 indicated that their first and most important source in the soils could be from erosion and runoffs of natural parent materials from the soil into the river watershed. These components were of lithologic (parent materials) and atmospheric origins rather than anthropogenic. According to (Ciszewski & Grygar, 2016), water erosion from urban soils and agricultural fields increased the concentration of metals in the receiving river sediments. This was because water erosion was a selective process, preferentially detaching and transporting clay and silt, which were regularly bound with heavy metals (Laceby et *al*., 2017). The soils in the Yaounde region were Ferralsols derived from the parent material of migmatitic gneisses (Ngon Ngon et *al*., 2009). As the lowland soils were slightly acidic, characterized by a low average pH (6.81), this favoured weathering with high levels of metal concentrations (Rahman et *al*., 2018). Since the lowlands were low-lying valley areas, water erosion from the watershed could bring large amounts of Al, Mn and Fe rich soil materials, explaining the natural toxicity of the soils in these elements (Rahman et *al*., 2018). The compatible siderophilic and chalcophilic behaviour of these elements (Ni, Cr, Co, and Zn) in iron minerals and their presence as lattice components in the associated heavy minerals were the likely factors that determined their migration characteristics as well as their enrichment and increased pollution potential in soils. The study conducted by Ekoa Bessa et *al*. (2021) on the mineralogy and geochemistry of the Ossa lake complex sediments in southern Cameroon showed that the heavy minerals associated with the sediments were quartz, zircon, rutile, goethite, gibbsite, feldspar, and accessory vivianite, and consisted of metals such as chromium, nickel, zinc, cadmium and cobalt. In addition, there were universities, industries, and market and aluminum fabrics located in the lowland areas that could have added wastes to the watersheds, contributing to this pollution.

The second component (PC2) formed with Cu and Pb presented that they came from similar sources. However, Pb and Cu also showed a moderate positive charge (0.492*), suggesting that the sources of Pb could be both natural and anthropogenic. After the metals released, their concordant migration and association with other contaminants by erosion and hydraulic transport processes could explain the geogenic source in lowland soils. Indeed, Edith-Etakah et *al*. (2017) stated that the geochemical mechanisms by which metals bind to finer-grained soil particles, causing moderate contamination, included adsorption, complexation, and coprecipitation, in a largely acidic environment. In addition, given that the Yaounde lowland soils studied were characterized by low pH, this could have resulted in the fixation and unavailability of soil nutrients such as phosphorus (Penn & Camberato, 2019). Therefore, the addition of phosphate fertilizer (Cong, 2017), resulting in greater accumulation of these metals in lowland soils and subsequently in sediments due to the selective process of water erosion. PC1 and PC2 together explained 81.32 % of the total variance, indicated that the lithogenic factor dominated the distribution of most metals considered in this study.

III.2.2.2. Water physico-chemical and heavy metal concentrations in lowland sites

The different physico-chemical parameters recorded in each of the three sites studied (4, 9 and 11) during the rainy season were lower than those recorded during the dry season, except for pH and Eh. However, the significant difference was not found between the physico-chemical parameters analyzed during both seasons ($p>0.05$) showing the effects of different lowland sites on water quality. Temperature is an important parameter of the aquatic environment because most biological and chemical parameters are dependent on it as well as mineralogical processes developped. Water temperature changed significantly between sampling sites during the rainy season (min 24.8 ±1.1 °C) compared to the dry season (max 28.83±1°C) where there was no significant change between sites. This can be explained by the influence of the local environment such as the geographical position of the lowland site, the size of the site, the shape and depth of the watershed. However, the temperature variations observed between the two seasons could be attributed to the natural climatic conditions prevailing in the study area. Thus, the T°C was within the WHO standards. The study conducted by Djuikom et *al*. (2009) on the physico-chemical quality of water in the Mfoundi river watershed in Yaounde, showed non-significant variations in temperature from one sampling point to another and observed values in the range of 20°C to 26.28°C. The authors explained that the non-significant change in water temperature in the Mfoundi watershed was a typical phenomenon of the tropics, where environmental conditions, and in particular temperature, remained relatively constant. In addition, another study conducted by Santsa et *al*. (2018) in the Menoua watershed in the West region of Cameroon, had recorded min and max temperature values (20°C and 25.20°C) during the rainy and dry seasons and found no significant difference on spatial variations. The discrepancies recorded could be due to the time difference between the different sites and sampling points.

The measurement of pH provides information on water quality and has been used to quantify the concentration of H^+ ions, which can be acidic or basic (Ravikumar et *al*., 2013). At all sites studied, the pH of water was slightly acidic to alkaline and within the normal range of pH of wastewater used for irrigation (6.5 - 8.4) (Anonymous 5, 2017) indicating normal development of flora and fauna in the lowland area. The slight acidity of water could be due to an input of humic and fulvic acids resulting from soil leaching by rainfall (Merhabi et *al*., 2019). As reported by Segbeaya, (2012), low water pH leads to acidification of the environment and promotes the release of metal complexes into the sediment. In both seasons, the high pH recorded in the lowland water could be attributed to high buffering capacity in the slow flow of water in the lowlands. Indeed, Kumar et *al*. (2020) had indicated that river water has a low buffering capacity when the flow rate is high, while the capacity decreases when the flow rate of river water is low. The high buffering capacity in our study could be due to the fact that ions derived from weathered bedrock and erosion are elevated in the surface waters (Grochowska, 2020), which resulted in an increase in the alkaline pH of the water. Indeed, the increase in pH at all lowland sites during the rainy season could be an indicator of chemical oxygen demand (COD) and ammonium (NH_4^+) oxidation via the nitrification reaction producing carbon dioxide (CO_2) by plants and inducing sediment acidification. This could explain the acidic pH observed in lowland soils, especially during the dry season. In the study conducted by Tchadanaye et *al*. (2017) in Western Mayo Kebbi (Chad), water pH values (6.91 to 7.53) were observed during the dry and rainy seasons due to leaching during rainfall. Another explanation for the difference in pH observed between each site could be due to the type of activities as hypothesized by Santsa et *al*. (2018); Kumar et *al*. (2020), the types of waste, leachate and other agropastoral and dumping activities located near water points or in streams in the catchment area probably affected the fluctuation of the water pH.

Electrical conductivity reflects the overall mineralization rate and provides information on salinity (Lakhili et *al*., 2015). Conductivity values were observed to be too low (216 to 502.33 μS.cm-1) in the different sites studied compared to the wastewater threshold values (3000 μS.cm-1). These low values indicated that the lowland water was less mineralized and explained the lower observed water salinity level (0.0 - 0.2 %). Rodier (2009) had reported that water was naturally saline if EC was between 1000 and 3000 μs/cm. The increase in conductivity values during the dry season could be due to the decomposition of organic matter and fertilizers in the soil. Therefore, Ali et *al*. (2020) showed that rooted aquatic plants tended to accumulate dissolved salts in wastewater with increasing time in association with the phyto-extraction process leading to pollutant extraction. Such values were reported in the Menoua watershed in the western region of Cameroon (39.5 to 459.75 μs/cm) by Santsa et *al*. (2018), surface waters of the Nkam River at different station points during the dry season (47, 2 μS/cm and 230. 25μs/cm) by Togue et *al*. (2017) and the surface waters of the Bétaré-Oya gold mining area (East-Cameroon) during dry and rainy periods (11.60 < EC < 122.10 μS/cm) by Rakotondrabe et *al*. (2017). All these authors linked the observed low conductivity values to the spatiotemporal evolution and the increase in ion contents related to the activity of pollutants in the studied areas. According to Madu et *al*. (2017), water quality parameters influence heavy metal levels and their accumulation in aquatic biota. In addition, Ma et *al*. (2016) had observed that among the physico-chemical parameters, pH of about 6.5-7 was favorable for the mobility and availability of metals in water and soil.

In this study, physico-chemical parameters varied significantly ($p<0.05$) between sites. Temperature and pH were the important parameters of the lowland environments because their variability influences most of the chemical parameters. The average water temperature at sites 9 (Mvan) and 4 (Mokolo-elobi) was higher 28.56±1.1 and 26.97±0.1 ºC respectively compared to 24.93±1.46 at site 11 (Atemengue pond Obili). This could be due to measurement and sampling time, geographical location or industrial effluent flow to the lowlands. Indeed, the Cameroon Brewery and the SOFAVIN industries (wine production) were located upstream where the hot wastewater from the machines was discharged directly into the Mfoundi River. Similarly, the Abiergue River at Site 4 (Mokolo-elobi) was surrounded by garbage, iron burning, metal and mechanical activities at the Mokolo market in Yaounde. Studies have reported changes in water temperature resulting from certain blockages in the rivers, leading to an increase in water temperature downstream (Kumar et *al*., 2020). In the current context of Cameroon, the lowlands suffer from the discharge of all types of degradable and non-degradable wastes (including plastic bottles) and climate change may have affected plants growing in the lowlands. Indeed, changes or increases in temperature may have had impacts on the wetland ecosystems (Wang et *al*., 2019).

Regarding, pH which provide water quality (acididic or basic), the high levels of the average pH value was observed at Mvan in site 9 (8.1±0.44) and Atemengue pond Obili. in site 11 (7.69±0.39) compared to Mokolo-elobi in site 4 (7.59±0.19). These high pH values were explained by the nature and characteristics of the environment and anthropogenic activities. Lowland sites were known to be anoxic areas where the CO_2 concentration was elevated and this also depended of the settlement of sediments. The alkalinity of pH may be due to high buffering capacity in slow flow of lowland water. According to Kumar et *al*. (2020), the pH and alkalinity increased when the buffering capacity of water was high due to the liberation of ions derived from weathered bedrock. Garbage, sediment and silt were quickly settle down to the river bottom in slow flowing streams. The fluctuation of pH in the differents sites could be due to the level of contaminants. Water alkalinity increase indicates high deterioration of water quality, which was influenced by various salts, wastewater, and organic fertilizers from agriculture. The negative values of Eh in the studied lowlands confirmed the oxidation process prevailing in these sites.

Electrical Conductivity is a good indicator of total salinity or total amount of dissolved solids in water. It reflects the rate of global mineralization and provides information on the salinity (Lakhili et *al*., 2015). Though it does not provide detailed information about the ionic composition in water, it can be used to determine the suitability of water for drinking and irrigation use. Electrical conductivity (EC) and Total dissolved solid (TDS) were higher in sites 9 and 4 than site 11. The low values of EC obtained at all sites (EC <600 µS/cm), indicated the lower presence of salt and inorganic materials in water. Studies on water quality pollution with heavy metals have reported the values of EC higher than 1000 µS/cm in wastewater used for irrigation (Nedjma et *al*., 2019; Kumar et *al*., 2020).

In the current study, metals in lowland water varied considerably with seasons. Among the metals studied, the concentration of Cd (0.0052 mg. L^{-1}), As (0.1705 mg. L^{-1}) and Co (0.1703 mg. L^{-1}) were higher than the WHO standard of 0.003, 0.01 and 0.05 mg/L, respectively. This was indicated that water at lowland sites was not safe for drinking and/or cooking. In the study carried out by Mambou et *al*. (2020) in Betare-Oya (eastern Cameroon), similar observations were made for Pb (0.018 mg. L^{-1}), Cr (0.02 mg. L^{-1}) and As (0.015 mg. L^{-1}) where the median values for these concentrations were above the thresholds values for water. In another study on the surface water quality assessment carried out by Rakotondrabe et *al*. (2017) in eastern Cameroon, Pb (100µg/L), Cr (60 µg/L), and Cd (50µg/L) were above the standard values. All of these studies showed variation in metals exceeding the WHO standard, which could depend on regions, sites and activities around the sites. Compared to these studies, the concentrations of As and Cd obtained showed that the highest values enlightened the degree of toxicity of these metals in the lowland areas.

Since the content of some elements such as Pb, As and Cd were higher than the WHO standards, this could indicate that their origins in lowland waters might have been influenced by other factors such as anthropogenic activities (domestic effluents and wastewater, industrial wastes, fertilizers, leachates, paints) and the residues from theses activities might have been leached. However, according to Cui et *al*. (2019), Król et *al*. (2020), the solubility of these heavy metals was strongly governed by pH through the precipitation of their oxides and hydroxides. Nevertheless, in the study area, most of the pH values tend towards neutral-alkaline, which does not allow the phenomenon of acid drainage but rather neutral-contaminated drainage. As stated by Huang et *al*. (2020), these phenomena may contribute to significant metal concentrations in these samples.

Several authors had attributed the highest metal contamination of water, to natural sources or human activities (Agoro et *al*., 2020; Kumar et *al*., 2020; Agoro et *al*., 2020; Merga et *al*., 2020; Zhou et *al*., 2020). Similarly, the alarming increase in concentrations of trace metals in Yaounde lowland waters, and for some elements exceeding threshold values, could present toxicity risks for consumption and on health. Indeed, the circulating surface water has been used downstream by the population for washing vehicles, motorcycle cabs, watering crops for direct consumption (salad, vegetables), for laundry, washing cars, bathing, and in some areas for bathing young people. In addition, the increase in heavy metal levels in lowland waters could be directly related to anthropogenic activities, especially the discharge of untreated effluent, which has also been identified as the main source of toxic metals in lowland areas (Ali et *al*., 2019). On the other hand, the physical appearance of these lowland waters had changed in color from blue to blackish. This could have been due to the heavy metal loading in these waters. Kumar et *al*. (2020) stated that the heavy pollution load of a riverine ecosystem deteriorates water both physically and biologically.

The average concentrations of the studied heavy metals were found in the decreasing order of: As>Co>Cd>Zn>Ni>Cu>Cr>Pb for both rainy and dry seasons. Statistical analysis showed no significant difference ($p>0.05$) in metal concentrations between sites. However, the change of seasons showed the great influence on the concentration of heavy metals in water. In fact, all metals, except Cu, showed higher values during the dry season than during the rainy season. The lower concentrations of toxic metals observed during the rainy season could be due to the dilution effect of water. Ali et *al*. (2018) in their study of the assessement of toxic metals and sediments of Pasur River in Bangladesh reported that the concentrations of metals in surface water varied with seasons, and the winter season had shown higher values than summer.

- **Multivariate statistical analysis of heavy metals in lowland water during the rainy and dry seasons.**

In this study, during the rainy season, significant negative correlations ($p<0.05$) were observed between pH and Pb, As and Co, which were explained by the fact that the increase of pH values negatively affected the concentration of these metals in water and led to the deterioration of these metals. Possible reasons for these results could be the recurrent flooding of lowlands and the dilution of water during rainfall. Indeed, pH would have had a negative influence on the mobility of Pb, As and Co, as low values were observed during this season. The strong positive correlation between Ni-Co (0.961**), Co-As (1,000**), As-Pb (1,000**) and Pb-Ni (0.961**) in lowland water indicated that their source of entry was runoff sediments from natural rock weathering areas. According to Dayioglu et *al*. (2018), Cui et *al*. (2019), the leaching of elements, especially hazardous elements, is mainly dependent on the environmental conditions (pH, temperature, reaction time, liquid to solid ratio). As reported by Mishra & Kumar (2021), the positive correlation coefficient between the heavy metals showed their current characteristics, mutual dependence and common input source. Furthermore, it was indicated by Mishra & Kumar (2021) that the Narmada River (India) received metal contaminated by wastewater from both anthropogenic and natural sources. According to their study on the estimation of physico-chemical characteristics and associated metal contamination risk in the Narmada River, India, they had found the highest Pearson correlation coefficients between Cu-Pb (0.998) and Zn-Cu (0.986), indicating that the wastewater in the river originated from electroplating industries.

On the other hand, during the dry season, a non-significant correlation between pH and heavy metals was observed, which indicated that pH was not the main factor affecting the mobility of the studied metals in lowland waters (Barzegar et *al*., 2016). Proshad et *al*. (2020), in their study on the appraisal of heavy metal toxicity in surface water of an urban river in vicinity of Bangladesh, was observed that the interrelated between pH and metals was insignificant. Therefore, when considering the relationships between the combinations, it exhibited significant positive associations, which indicated that the parameters were interrelated and might originate from the same sources. However, a negative correlation was shown between EC, TDS and Pb, As, Ni, Cd and Co, which explained that these factors (parameters) were the ones that most affected the mobility of metals in Yaounde lowland waters. The strong positive correlations between the metals themselves was explained As-Cd (1.000**), Cd-Co (1.000**), Co-As (1.000**), Cu-Ni (0.993**), Zn-Cd (0.834) and Zn-Cd (0.893), indicated common sources of these metals in water and identical behavior of their influence in lowland waters. Indeed, Mishra & Kumar (2021) stated that the positive correlation coefficient between heavy metals demonstrated their actual characteristics, mutual dependence and common input source.

Principal component analysis (PCA) identified two significant components for the identification of heavy metal sources in lowland waters of the three sites. The first principal component (PC1), which contained 69.08% (rainy season) and 54.06% (dry season) of the calculated variance, showed a strong positive loading for Ni, As, Co and Pb, but moderate for Zn. This was explained by the anthropogenic source of these metals in the water during the rainy season. In addition, this was also supported by the very strong positive significant correlation observed between As, Pb and Co (1,000**), which were mainly from fertilizers, limestone or manures. Since corrosive materials resulted from waste deposits on riverbanks may wear out over time. These could have contributed to the positive loading of Ni (Achi et *al*., 2021). Indeed, As was positively related to Ni ($r = 0.961^{**}$), indicated that in addition to the anthrogenic source of these two elements, As came from bedrock materials. A similar observation was made by Huang et *al*. (2020) in their study on the distribution, sources, and ecological risk assessment of heavy metals in the Huixian wetland in southern China, which observed a strong positive loading for As and Ni, but moderate for Cr. These authors explained that chromium and nickel were known to be mutually associated with several rock types. According to Benhaddya et *al*. (2020), Ni and Pb levels were influenced by agricultural sources from wetland sites which was due to their location in the watershed that received agricultural drainage. Agricultural activities, atmospheric transport, waste incineration around the lowlands contributed to the observed elevated levels of Pb and As, as these metals could be presented as impurities in fertilizers and in metal-based insecticides, fungicides, compost and manure. However, Palansooriya et *al*. (2020) indicated that in agricultural areas, the profiles of potentially toxic soil elements were more closely related to lithogenic sources. Almost the same elements were reported in the study conducted by Ali et *al*. (2018) who observed in PC1 metals as Cd, As and Cr in Bangladesh water, while Proshad et *al*. (2020) in the surface water observed Cu, As, Cd and Pb in PC1 during winter. All these authors linked metals in PC1 to anthropogenic activities, which were mainly from agricultural activities such as wastewater irrigation and fertilizer application, industrial metal discharges such as wires, petrochemicals and automobile batteries. The second group (PC2) showed 30.92% of variance with high Cu loadings mainly from anthropogenic and lithogenic/geogenic sources due to weathering and/or leaching of fine particles of this element from sulfide rocks in the lowland system. This was also observed with the positive correlation with As, Ni, Co, Pb and negative correlation with Zn.

During the dry season, As, Co, Cd contributed the most in PC1, which indicated mainly anthropogenic and natural sources of metals during this season. This was also supported by the strong positive significant correlation observed between As, Cd and Co (1.000**). Indeed, these metals were commonly found in wastewater from agricultural areas and fertilizer industries via the leaching process (Wang et *al*., 2017; Garba & Abubakar, 2018; Gao et *al*., 2020). The application of As, Cd and Co pesticides/herbicides during agricultural activities in the lowlands of Mokolo-elobi (site 4), University of Yaounde I (site 11), and reagents used in laboratories (UYI) explained the anthropogenic sources of these elements. Alternatively, their inputs to lowland waters could be attributed to natural hydrogeochemical processes. Proshad et *al*. (2020) in their study on the assessment of heavy metal toxicity in surface water of an urban river near industrial areas of Bangladesh, observed that during the summer season, Cr, Ni, Cu and As contributed the most in PC1. The authors indicated that these elements were mainly from anthropogenic inputs such as chromium salt use for tanning activities, pesticides/herbicides, especially for As (Hassan, 2019; Huang et *al*., 2020). Taking into account that Cd was an extremely toxic element with high ecological risks, the very high and significant correlation with As and Co in this study confirms this. The presence of these pollutants was probably due to the long-term use of phosphate fertilizers, reagents or deposits, in addition to the high natural background in the lowlands. According to Proshad et *al*. (2020), Cd was derived from weathering and/or leaching of fine grains and the sorption and desorption behavior of this element in surface water. The second group (PC2) with high loadings of Pb, Cu, Ni, Cr and Zn came from both anthropogenic and geogenic sources. The negative loading of Cr (r= - 0.836**) and Pb (r= - 0.855**) during the dry season could characterize the geogenic process. The sources of Cu could be weathering and/or leaching of sulfide-bearing rocks, which explained the geogenic sources of this element in water (Benhaddya et *al*. 2020), Ni concentration levels in water were relatively low, with a maximum of 0.014±0.0 at Mokolo-elobi (site 4). According to Benhaddya et *al*. (2020), nickel could form insoluble compounds (hydroxides or sulfides) when the pH was alkalined (pH >9). In addition, Genchi et *al*. (2020) explained that Ni enters surface waters via a waste stream (anthropogenic sources) and can either dissolve or be complexed with inorganic ligands. Nickel could also be readily desorbed from clay. In addition, the high inorganic salt content of the water provided competition to other cationic salts for available sorption sites (Voisin et *al*., 2017), and the formation of chloride complexes potentially inhibited nickel sorption (Klapper et *al*., 2017; Sisso et *al*., 2020). However, naturally occurring trace levels of Cr could be detoxified naturally by reduction and precipitation (Benítez, 2019). The high levels of Ni and Cr in surface waters could be attributed to corrosion in stainless steel wells (Cardoso et *al*., 2019; Carranza & Rodríguez, 2017). Zn was used as an anti-corrosion agent when applied to iron pipes to protect them from corrosion, and these galvanized pipes were used during borehole construction (Khan et *al*., 2020; Abdelsattar et *al*., 2020) and to protect stream easements. The high Zn levels for these samples in PC2 could be attributed to corrosion of the parts at Mokolo-elobi (site 4). The elements (Cu and Zn) in water in natural soils were asserted to show a close geochemical dependence as the iron family (Huang et *al*., 2020), which was presented again in the present results with a negative correlation coefficient of r = -0.772*. Indeed, Huang et *al*. (2020) explained that commercial phosphate fertilizers contained small amounts of different elements (such as Zn, Cd and Pb) and they came from their raw materials as compared to other inorganic fertilizers.

Among the groups during both seasons, one group PC1 showed almost similar loadings of As, Co, Pb, Ni, and Zn (rainy season) and As, Co, Cd (dry season) in water, indicated that these loadings were mainly due to anthropogenic activities and geogenic sources. However, when comparing water and soil, PC1 revealed almost similar loadings of Cr, Cd, Co, Zn, Pb in water and Cr, Cd, Co, Zn, As and Ni in soil, indicating that they were mainly contributed by anthropogenic activities. The study conducted by Ali et *al*. (2018) on the assessment of toxic metals in water and sediment of Pasur River in Bangladesh showed that among the two groups, one group revealed similar loadings of Cd, As and Cr in water and Cr, Cd, As and Pb in sediments, indicating that these elements were mainly due to anthropogenic activities. The similarity observed between some elements found in both water and soils could indicate the link and complementarity between these two systems and the increase in the level of metals in the environment due to anthropogenic activities. However, in this study, PCA revealed that the distribution of the same type of heavy metals in water and soil was not similar, which could be due to the emission of toxic metals in the environment and addition through water. In addition, according to Popoola et *al*. (2018), Müller et *al*. (2020) deposition of atmospheric particles released from automobile emissions could contribute to the presence of these metals in the urban areas from where the water and soil samples were collected.

Cluster analysis (CA) divided the heavy metals in water into several groups, which showed that during the rainy season, Pb, Co, As and Ni formed a cluster that showed the anthropogenic and geogenic sources where Cu and Zn were linked to this cluster with a large binding distance. During the dry season, Co, As and Cd formed a cluster where Zn, Cu, Ni, Cr and Pb were linked to this cluster with a large binding distance. Compared to the study carried out by Proshad et *al*. (2020), which observed that during winter, the clusters were formed by Ni, Cu, As, Cd, Pb and Cr. Then, during summer, they observed that Cr, Ni, Cd, and Pb formed a cluster where Cu and As were bound with a large distance. The differences and similarities observed among the homogeneous clusters formed by trace metals in each of these studies could reflect the influences of tributaries, types of industries, and vehicle exhaust, as well as the agricultural activities that directly surround the sites in these regions (Wang et *al*., 2017).

Heavy Metal Toxicity Index (HMTL) gave an indication of the level of treatment required to treat wastewater to acceptable levels for use. The average HMTL values (141561.94 µg/L) observed during the dry season (85579.49 - 177426.11 µg/L) were very high compared to their average (43116.73 µg/L) during the rainy season (28737.12 to 60203.14 µg/L) at all three sites. The increase in toxic load during the dry season could be explained by environmental conditions and factors affecting lowland dynamics. Therefore, factors including lowland size and morphology, water depth and volume or flow during each season, and watershed shape could influence metal loading in water. In addition, physico-chemical parameters such as pH and temperature could influence heavy metal loading. Indeed, temperature varied from 25.07±2.0 to 28.83±2.06, and pH 7.35±0.28 to 7.93±0.3 during the dry season in the different sites, then from 24.81±1.1 to 28.3±0.72, and pH 7.58±0.0 to 8.25±0.5 during the rainy season. These temperature and pH values were favorable for the release of metals in water and consequently increase the pollutant load. The studies carried out by Li et *al*. (2013), Król et *al*. (2020) on the effects of pH, temperature, dissolved oxygen, and water flow on the release of heavy metals showed that, the release rates of heavy metals were more affected under low pH (4 - 7) than high pH (8 - 10) conditions and at high temperature (30 - 35 °C). These authors showed that the release rates of metals were increased more rapidly than at low temperature. In addition, the authors observed that the release of Zn, Cu, Cr, and Pb appeared to increase under aerobic conditions; adsorption and release of Cd occurred under anaerobic conditions, flow rate affected the release of Zn, Pb, Cr, and slightly Cu and Cd. Proshad et *al*. (2020) in the study carried out on the assessment of heavy metal toxicity in surface water on an urban river near the industrial areas of Bangladesh, observed the same trends in the variation of HMTL. The authors observed during summer (average: 36607.3 mg/L, min 19699.6 mg/L and max: 48854.7 mg/L), than (average: 42208.2 mg/L, min: 22489.2 and max: 57781.1 mg/L) during winter for six analyzed metals Cr, Ni, Cu, As, Cd and Pb. Compared to this study, the HMTL observed was very low for the current study. This could be explained by the types of activities around the different sites, their intensities in the areas and the level of industrialization of the regions. Furthermore, the reasons for the difference in contamination and elevated metal load in water across seasons was explained by Duncan et *al*. (2018) as the leaching of upper sediments by heavy rainfall and high runoffs during the wet season and low flow during the dry season which would facilitate the precipitation and accumulation process.

Heavy metal evaluation index (HEI) showed the overall water quality and was classified in terms of pollution level in the three lowland sites. During the rainy season, the average value was 1.6, while it was 3.98 during the dry season. The HEI was less than 10 at all three Yaounde lowland sites, indicating low metal pollution in lowland waters. However, Proshad et *al*. (2020) observed in the surface waters of urban areas of Bangladesh the low level of pollution (HEI <10) and the values of 1.15 during winter and 0.92 during summer. The variation of pollution index during seasons in these studies could be explained by climate characteristics and natural factors related to each studied sites. On the other hand, the seasons were an important factor in the assessment of metal pollution, because their fluctuation balanced the average pollution in the studied areas. Maurya et *al*. (2017) explained that environmental factors such as climate (seasonal variations and temperature), drainage wastewater affected the physical and chemical characteristics of water and its quality. In addition, seasonal and spatial distributions of heavy metals means that prevention and mitigation measures should be targeted taking these variations into account (Yao et *al*., 2014). In another study by Benhaddya et *al*. (2020), on the assessment of heavy metal pollution in surface and groundwater systems in the Oued Righ region (Algeria), the heavy metal evaluation index (HEI) values ranged from 2.33 to 7.07, with an average value of 4.03 for surface water. All of these studies presented a low pollution index, however with values that progressively changed according to the sites.

Ecological risk index (ERI) followed the same trend in both seasons. They were lower for all the sites studied but with the highest contribution in water pollution of Cu (rainy season), Ni and Pb (dry season). In the study carried out by Proshad et *al*. (2020), the ERI was moderate and low for both seasons with the highest contribution of As in water pollution. Therefore, according to Li et *al*. (2018), Chiou & Hsu (2019), excess Cu would cause severe alterations in plant metabolic functions, especially due to excessive production of reactive oxygen species (ROS), leading to disturbances in vital cellular processes such as photosynthesis, respiration, and metabolism of important enzyme functions. Furthermore, Huang et *al*. (2020) explained that, the reduction in CO_2 assimilation by leaves induced by copper toxicity could be caused by a decrease in the abundance of proteins related to the photosynthetic electron transport chain (PETC) and CO_2 assimilation. Indeed, ingestion of high levels of soluble copper salts would cause acute gastrointestinal symptoms in humans and, in frequent cases, liver toxicity in individuals sensitive to repeated exposure (Taylor et *al*., 2020). On the other hand, Pb would cause toxic effects (physiological, morphological and biochemical) in living organisms at very low concentrations when exposure occurs during a long period. In plants, Pb toxicity would be characterized by an alteration of chlorophyll (Chl a) production, cell division, root elongation, lamellar organization in the chloroplast, plant growth, plant stress, seed germination, seedling development and transpiration. Pb absorbed in the gut was then transported to soft tissues, e.g., liver, kidney, and bone tissue, where it accumulated over time (Guo et *al*., 2018; Kumar et *al*., 2020). However, nickel at high levels would impair metabolic activities in plants by inhibiting enzyme activity, photosynthetic electron transport, and chlorophyll biosynthesis. Contact with nickel would cause various secondary effects on human health, such as allergy, cardiovascular and renal diseases, pulmonary fibrosis, lung and nasal cancer. Although the molecular mechanisms of nickel-induced toxicity were not yet clear, mitochondrial dysfunction and oxidative stress were thought to play a primary and crucial role in the toxicity of this metal (Genchi et *al*., 2020).

III.2.2.3. Heavy metal contents in *Echinochloa pyramidalis, Pennisetum purpureum* and *Commelina benghalensis* from lowland of Mokolo-elobi (site 4), Mvan (site 9) and Atemengue pond Obili (site 11) of Yaounde during the rainy and dry seasons

Heavy metals analyzed in *E. pyramidalis*, *P. purpureum* and *C. benghalensis* in the three sites during the rainy and dry seasons showed higher concentrations of Pb, Cr, Ni, Zn and Cu exceeding WHO standard values except As in plants. This indicated that these species possessed the ability to accumulate metals in their tissues (shoots and roots). As some metals in plants were beyond the toxicity range, they revealed high levels of tolerance against the toxicity of the selected metals. The concentration range of Zn in the plants showed high variability between 67.82 and 252.62 $\mu g.g^{-1}$ for the three plants analyzed, indicating high accumulation by the plants. According to Chandra et *al*. (2018), plants with a tendency to accumulate >1000 mg/kg were known to be hyperaccumulators. However, the study by Parihar et *al*. (2020) on bioaccumulation potential of indigenous plants for phytoremediation of heavy metals in rural areas of Shaheed Bhagat Singh Nagar, Punjab (India) showed that the concentrations of Cd, Cu, Cr, Fe and Mn were within the standards except for Co (5.950 - 21.20 mg/kg) and Zn (16.04 - 113.8 mg/kg). The high levels of Zn presented in these two studies could be explained by the nature of this micronutrient and its importance in plant metabolism. Indeed, Zn was an essential micronutrient for plant growth and development. According to Balafrej et *al*. (2020), plants could exhibit several morphological, physiological and biochemical adaptations resulting from the activation of molecular mechanisms of Zn hyperaccumulation. Thus, the three species studied showed high capacities to accumulate Zn. In addition, the increase in Cd level during the dry season exceeded the limit in all species with the maximum value (0.41 ± 0.38) µg.g-1 in *C. benghalensis*. This was explained by the bioavailability and transpiration of plants during this season. Indeed, according to Sheoran et *al*. (2016), the transpiration rate was dependent on the climate. As a result, during the rainy season, plants absorb little water when flooded, compared to the dry season when plants absorb and transpire a large amount of water to facilitate photosynthesis. The variations in metal concentration observed in plant species at the studied sites and between seasons were in agreement with the results obtained by Eid et *al*. (2020), Kumar et *al*. (2019b) who observed the influence of seasons on heavy metal concentrations in different plant species. Indeed, metal uptake was dependent on various abiotic and biotic factors and was also influenced by metal antagonism and competition. According to Dinu et *al*. (2020), the presence of antagonistic couples such as Ni-Co, Cr-Zn, Zn-Pb, Cd-Zn, and Cu-Zn induced different developmental behaviors in plants. Thus, the presence of some metals inhibited the uptake of others or increases/decreases their toxicity (Gajic et *al*., 2018).

Comparing the three plant species, the results showed that *C. benghalensis* accumulated the highest concentrations of Zn, followed by *P. purpureum* and *E. pyramidalis*. This can be due to the plants morphology, their physiological stage and their capabilities to uptake metals from soil. According to Parihar et *al*. (2020), the uptake of toxic elements by plants simulated the mechanisms involving translocation and storage of micronutrients. Therefore, Kassaye et *al.* (2016) explained that the variation in heavy metal accumulation could be due to the different growth forms of these species. In addition, heavy metal concentrations in aquatic macrophytes changed significantly depending on the type of species and their tissues as well as the type of heavy metals (Fu et *al*., 2019).

In this study, plant tissues significantly affected the accumulation of all studied metals, except As, in each plant. The results showed that the metals were accumulated more in the roots than in the shoots (leaves and stems). The decrease order in accumulation of Cu, Cr, Ni, Pb, and Co in plants was similar to those obtained by Kamari et *al*. (2017), Galal et *al*. (2019), Eid et *al*. (2020) who found that species accumulated higher concentrations of heavy metals in their roots than in their shoots. The exception was for Cd and Zn, which showed a different order of accumulation in plant organs. According to Shahid et *al*. (2017), the accumulation of heavy metals in plant roots was an exclusion strategy because the root was not a photosynthetic organ, this might increase plant tolerance to toxic concentrations of these metals. In aquatic macrophytes, the heavy metal compartmentalization strategy was common, and plants sequestered high concentrations of metals in their underground tissues to protect themselves from the harmful effects of metal toxicity (Bonanno et *al*., 2017). Nevertheless, roots tended to have higher concentrations of metals compared to stems and leaves. Therefore, they acted as a barrier for metal translocation and protected the stems and other plant parts from metal contamination (Shahid et *al*., 2017; Sulaiman & Hamzah, 2018). Similarly, Rezania et *al*. (2019) found that *Phragmites australis* could be used to immobilize certain metals, such as Cd and Pb, and store them in its underground tissues.

Correlation matrices provided knowledge about heavy metal sources in plant parts related to water and soil (Wang et *al*., 2017; Wang et *al*., 2019). A positive and significant correlation was found between almost all metals in plant parts of *E. pyramidalis*, *P. purpureum*, and *C. benghalensis* and the inter association with metals in soil and water of lowland sites. This could be the result of the similarity of metal sources in water and soil, which were bioavailable for uptake by different plant organs (roots and shoots). According to El-amier et *al*. (2018), significant correlations of metal contents in plants or plant parts and other matrices such as soils and waters demonstrated that metal accumulation by plants reflected the temporal fluctuation of elements in water and soil. Furthermore, Mbale Ngama et *al*. (2019) explained that the observed positive correlation between the same trace elements was attributed to local hydrodynamic forces that could be related to the accommodation of these elements by the same ferromagnesium minerals (Ni, Zn, Cu, and Co). However, Cd and As showed a weak negative correlation with the other metals in the three plant species, soils, and water during both seasons. This was attributed to the difference on the parts of uptake and accumulation of these metals by plants. In the study conducted by Fu et *al*. (2019), on the uptake and transport of heavy metals by native wild plants: implications for phytoremediation and restoration, Cd was taken up differently by plants. For example, for some species, leafy plants accumulated the highest concentration of Cd, followed by rooted plants and grains (Khan et *al*., 2018).

III.2.3. Remediation performance of plant species

Translocation factor (TF), mobility ratio (MR), and bioaccumulation factor (BAF) were used to estimate the phytoremediation potential of plants (Eid et *al*., 2019). The study showed that bioaccumulation of the plant species studied was influenced by seasonal changes. Zinc was the most extracted metal among all others. This was explained by the fact that Zn was an essential element in enzymatic and photosynthetic processes for plant growth. The translocation factor indicated the ability of plants to translocate the pollutant from the roots to the aerial parts of the plant. *C. benghalensis* exhibited the high translocation values (TF>1) of Zn, Cu and Cd and *P. purpureum* high TF of Cu, Cd and Ni during the rainy season. However, during the dry season, *E. pyramidalis* showed high transfer values of Ni and Cr, while *P. purpureum* showed high tranfer for Pb. This indicated that, the metals were accumulated in the shoots directly after uptake by the roots. The difference in metal uptake by plants could be due to the type of plants, their root morphology and characteristics, the type of metals and the ability of each plant to withstand stress conditions. According to Parihar et *al*. (2020), the uptake of toxic elements by plants simulated mechanisms involving translocation and storage of micronutrients. The study carried out by Eid et *al*. (2020) on the phytoremediation of heavy metals by four aquatic macrophytes (*Eichhornia crassipes*, *Ludwigia stolonifera*, *Echinochloa stagnina*, and *Phragmites australis*) in Egypt, showed that translocation factors (TFs) were less than 1 for most of the heavy metals, except for Cd in *E. stagnina* and *L. stolonifera*, and Ni in *E. stagnina*. Futhermore, Usman et *al*. (2019) explained that the transport of elements from plant roots to shoots considered highly efficient if TF > 1, revealed the existence of a well-organized and better metal transport system by plants.

High mobility ratio (MR) was observed for Cd in *E. pyramidalis* (2.74), *P. purpureum* (7) and *C. benghalensis* (24.42) during the rainy season, in addition to Cu (1.23) and Zn (1.08) which were in *C. benghalensis*. During the dry season, *C. benghalensis* and *E. pyramidalis* showed higher MR values for Cd uptake, 31.15 and 16.23 in addition to Zn and Cu in *C. benghalensis* only. These results showed that cadmium is a highly mobile metal compared to other studied metals like Pb, Cr, Co, As and Ni in different plants. This was because metals have different mobility and can be transported from roots to shoots in different ways. However, Kandziora-Ciupa et *al*. (2017) in their study on heavy metal bioaccumulation and ecophysiological responses to heavy metal stress in *Vaccinium myrtillus* and *Vaccinium vitis*-

idaea, observed a very high MR for Mn in *V. myrtillus* and *V.vitis-idaea*, explaining that Mn was used for beneficial purposes for these plants.

The bioaccumulation factor (BAF) indicated the efficiency of plants to accumulate the pollutant in its tissues (roots, stems and leaves). *C. benghalensis* during the rainy season showed the highest BAF value for Cd in roots (23.78), stems (24.29), while *E. pyramidalis* showed Cd in roots (2.768), and stems (1.956). However, these metals were higher in stems than in roots during the dry season. This indicated that Cd was more soluble and bioavailable to roots than the other metals Pb, Cr, Co, As, Cu, Ni and Zn in soils. According to Hasan et *al*. (2009), Haider et *al*. (2021) the bioavailability of cadmium in soil depends on pH, redox potential, temperature, and concentration in relation to the presence of other elements. Therefore, changes in the electrochemical potential of cytosolic Cd^{+2} and root apoplast were the main mechanisms controlling Cd uptake across the plasma membrane of root cells (Bali et *al*., 2020; Abedi & Mojiri, 2020). Similar observations were showed by Usman et *al*. (2019), Parihar et *al*. (2020), who observed the highest BAF values of cadmium in different plant species. *E. pyramidalis* and *P. purpureum* also showed higher BAF values for Cd in sites 9 and 11 compared to other metals, despite the less Cd-contaminated soils observed in all three sites. In addition, cadmium uptake across the plasma membrane of root cells was regulated by the electrochemical potential difference between Cd activity in the cytosol and in the apoplasts of roots. Thus, the membrane potential gave sufficient energy to drive Cd uptake even at a very low Cd concentration (Haider et *al*., 2021). In roots, Cd absorption may occur as inorganic complexes (i.e., $Cd^{2+}SO_4$, $CdCl^+$, and $CdCl_2$), or organic forms (i.e., complexes of phytometallophore), (Kubier & Pichler, 2019). In this study, almost all roots and stems of all plants studied were Cd phytoaccumulators. However, the roots and shoots of *C. benghalensis* showed relatively high and low BAF values for copper at site 9, even in the presence of a significant concentration of copper in the soil at the same site. This variation in the same plant at different sites could be due to the availability of varying forms (water soluble, exchangeable, inorganically bound, organically bound, oxide bound, and residual) of copper to plant species for uptake (Kasowska et al., 2018). All plant species acted as Cd phytoaccumulators in roots and shoots. In addition, *C. benghalensis* accumulated Cu, Co, and Zn in roots and shoots, while *E. pyramidalis* accumulated Ni and Zn respectively in roots and shoots in lowland sites during both seasons, reason being that they illustrated BAF values greater than 1. The efficiency of metal accumulation in plant tissues differred considerably among elements, plant species and organs. Cu and Co were accumulated mainly in the roots of *C. benghalensis,* whereas Ni was accumulated especially in the roots of *E. pyramidalis*. Both species behaved as Cu, Co, and Ni excluders, respectively, due to the restriction of heavy metal transport from roots to shoots (Stefanowicz et *al*., 2016). According to Huang et *al*. (2020), the type of exposure (root or shoot) could have different effects on metal compartmentalization (tissue and cell-level distribution) in plants and consequently metal bioavailability and toxicity. In the case of heavy metal uptake by plant roots, most of the absorbed metals, especially Pb (about 95% or even more), were sequestered in root cells, with limited translocation to aerial tissues, unless the plant was chelated or hyperaccumulative (Shahid et *al*., 2016) or assisted by microbes (Ojuederie & Babalola, 2017).

Results analysis for TF and BAF in this study showed that *C. benghalensis* had the potential to be used as a phytoextractor of Zn, Cu and Cd as the values of BAF and TF were observed greater than 1 (BAF and TF>1), *E. pyramidalis* as a phytoextractor of Cd and Ni, and *P. purpureum* as a phytoextractor of Cd. According to Bello et *al*. (2018), the plant species with a bioaccumulation factor (BAF) greater than 1 and a translocation factor (TF) less than 1 for a particular metal had the phytostabilization capacity. In the present study, some of the plants in response to heavy metals in their respective sampling sites indicated TF < 1 revealing their potential use as phytostabilizers by limiting the movement of heavy metals in roots. Among the three plant species, multi-metal phytostabilization ability was observed in all plants for Co and As. *C. benghalensis* was found to be a phytostabilizer of Ni, as well as *E. pyramidalis* for Cd.

The metal accumulation index gave the overall performance of plant species to accumulate metals according to its deviation in metal uptake. The highest MAI value was observed in shoots (47.73) and roots (47.55) of *P. purpureum* in site 9 during the rainy season. Whereas, during the dry season, *E. pyramidalis* showed the highest value in roots (19.20) and shoots (18.49) in site 11. The minimum MAI value was found in the shoots of *P. purpureum* 1.04 in site 4 during the rainy season, while it was 1.36 in the shoots of *E. pyramidalis* during the dry season. The difference found in metal accumulation during both seasons was probably due to meteorological and natural conditions such as temperature, flooding, weathering and environmental variations (soil properties, water characteristics). In addition, the current study takes into account soil and water pollution, since the shoots and roots of the studied species were in direct contact with these components, as well as the ambient air in lowlands. In addition, the plants, which showed the highest MAI value, were both from the Poaceae family. This indicated that herbaceous species of Poaceae family such as (*E. pyramidalis* and *P. purpureum*) showed high metal accumulation by growing in contaminated environments. However, Parihar et *al*. (2020) on bioaccumulation potential of indigenous plants for phytoremediation of heavy metals in rural areas of Shaheed Bhagat Singh Nagar, Punjab (India) observed MAI value of

27.99 in roots of *Morus indica*. However, previously, Nadgóska-Socha et *al*. (2016) on *Robinia pseudoacacia* and *Melandrium album* in trace elements biomonitoring and air pollution tolerance index study showed the highest MAI value in the roots of *M. album* (26.4) and the lowest in *R. pseudoacacia* (7.98). Moreover, in a study conducted by El-amier et *al*. (2018), on the potential of macrophytes for the removal of heavy metals from the aquatic ecosystem, in Egypt: using the metal accumulation index (MAI), the maximum average values of MAI were found in the roots of *P. australis* (36.19). The authors observed the minimum of MAI in *E. stagnina* (11.50 and 13.43) during the summer and winter seasons (15.80 and 16.95). Compared to these studies, the present study shows the highest value of MAI in plant roots, which could be due to the nature of lowland plants, their characteristics to resist metal pollution and the conditions prevailing in each of the studied sites. On the other hand, Parihar et *al*. (2020) explained that the high efficiency of native plant species to accumulate metals was due to the subhumid to semi-arid environment and atmospheric chemistry prevailing in the local areas studied. Furthermore, Hu et *al*. (2014) suggested that herbaceous plants with higher MAI values should be used as barriers between contaminated and non-contaminated areas.

Compared to the roots, 18% of the shoots (*E. pyramidalis* and *P. purpureum*) showed elevated MAI values at the different sites in the two different seasons. In this case, since the shoots were not in direct contact with the soils, the accumulation of heavy metals could be attributed to the aerial parts (leaves) via foliar uptake of heavy metals. Thus, leaves rapidly absorb a significant amount of metals that were deposited on their surface as dry aerosol particles (Hu et *al*., 2014; Kleckerová & Dočekalová, 2014). Indeed, Shahid et *al*. (2017) explained that metals could be absorbed and stored in leaf tissues with a small portion (<1%) transported to root tissues. As a result, the leaf parenchyma would contain most of the foliar applied metals. Thus, foliar uptake should not be neglected in the lowlands since they were located in urban areas and vehicles and motorcycles crossed the majority of sites.

The comprehensive bioconcentration index (CBCI) revealed the overall performance of plants in terms of bioaccumulation of multiple metals to assess their phytoremediation capacity (Zhao et *al*., 2014). This index synthesized the accumulation results of several metal pollution factors into one relative measurement index. By comparing the CBCI value of each of the studied species, the accumulation capacities of multiple metals were evaluated. The CBCI values showed that *P. purpureum* had the highest multi-metal accumulation capacity (43.5) in both seasons. However, *E. pyramidalis* and *P. purpureum* were the best multi-metal accumulators during the rainy season, while *C. benghalensis* was the best accumulator during the dry season at the lowland sites. The study carried out by Parihar et *al*. (2020) on the bioaccumulation potential of plants in Punjab (India), showed lower multi-accumulation capacity of heavy metals by plants (herbs) such as *Cannabis sativa* L. (CBCI 0.396), *Medicago polymorpha* L. (CBCI 0.328), and *Amaranthus spp*. (CBCI 0.474) compared to trees. In another study conducted by (Zhao et *al*., 2014), on the evaluation of heavy metal accumulation and application of a global bio-concentration index for woody species at contaminated sites in Hunan, China, showed for multi-metal accumulation capacity *B. papyrifera* (CBCI 2.93), *Amorpha fruticosa* L. (CBCI 2.72) and *Lagerstroemia indica* L. (CBCI 2.53).

Compared to these studies, the current study shows the highest CBCI values and this could be due to the fact that these species have short cycles and are highly invasive, colonizing and characterising highly eutrophied environments. These high pollutant loads have granted them tolerance capacities of metallic pollutants. Indeed, the large variations of metal and BAF concentration within the same species on different sites confirmed this point of view (Table XXVII). The present results would complement the information on the herbaceous families and show that in the lowlands of the sub-Saharan region, *E. pyramidalis*, *P. purpureum* and *C. benghalensis* were found to be the best bioaccumulators of the heavy metals studied.

The different correlations observed between each plant tissue and the parameters considered during this study in the three lowland sites gave a global overview of the pollution tolerance of each specie. Thus, *E. pyramidalis*, *P. purpureum* and *C. benghalensis* were able to tolerate metals in soils and waters and contributed to their mitigation. Therefore, they were not able to reduce the ecological risk of metals on lowland surface waters. Following these results and according to several authors cited in this work (Zhao et *al*., 2014; Kandziora-Ciupa et *al*., 2017; El-amier et *al*., 2018; Bello et *al*., 2018; Eid et *al*., 2019; Usman et *al*., 2019; Rezania et *al*., 2019; Eid et *al*., 2020; Parihar et *al*., 2020), *C. benghalensis*, *E. pyramidalis* and *P. purpureum* could be described as useful for phytoremediation of heavy metals in lowland areas. Moreover, *C. benghalensis* was the most effective followed by *E. pyramidalis* than *P. purpureum*.

CHAPTER IV. CONCLUSION, PERSPECTIVES AND RECOMMENDATIONS

IV.1. Conclusion

The aim of this study was to evaluate the bioaccumulation capacities of heavy metals in some plant species of the lowlands of Yaounde (Cameroon). Based on the results obtained in this study, the following conclusions were drawn:

- Out of an average of 164 plant species identified during the dry and rainy seasons, 20.6% showed suitability for phytoremediation of heavy metals in lowlands and more than ten species (*Echinochloa pyramidalis*, *Ipomoea aquatica*, *Commelina benghalensis*, *Pennisetum purpureum*, *Setaria barbata*, *Panicum maximum*, *Ipomoea batatas*, *Alternanthera sessilis*, *Alchornea cordifolia*, *Cynodon dactylon*) could be potentially effective in attempts to clean-up soil and water contaminated by heavy metals. Among them, three of the major plant species : *E. pyramidalis*, *P. purpureum* and *C. benghalensis* were selected for the metal accumulation potential assessment;
- the concentrations of Cr, Ni and Co in the lowland soils were higher compared to the WHO limits. Geo-accumulation index values indicated that soils were heavily contaminated with Cr and moderately with Cu from anthropogenic sources. Nemerow integrated pollution index (IPI) values showed that soils were polluted by heavy metals. Potential ecological risks (E^i_r) of toxic metals indicated a slight level of ecological risk, thus Cr and Cu contributed the most in the pollution of lowlands;
- Cd, As and Co in water were above WHO limits. Heavy metal toxicity load indicated low contamination of toxic metals in water, and the evaluation index specified low pollution of lowland waters and low ecological risks of metal in water of lowlands;
- heavy metal concentrations in *E. pyramidalis*, *P. purpureum* and *C. benghalensis* were all above the allowable limit except for arsenic. *C. benghalensis* presented the highest accumulation of Zn. Therefore, Cd concentrated in stems and roots and Zn in roots and stems were accumulated differently in plant organs than other metals;
- *C. benghalensis* showed the potential to be used as phytoextractor of Zn, Cu and Cd, *E. pyramidalis* as phytoextractor of Cd and Ni, and *P. purpureum* as phytoextrator of Cd, and they were all shown as phytostabilizators of Co and As. The current study revealed a high accumulation of metals in the Poaceae than the Commelinaceae family.

IV.2. Perspectives

In order to increase the knowledge related to the use of plants for soil and water decontamination, the implementation of phytoremediation techniques, the shortcomings and challenges of lowland management in Cameroon, this study could be completed by:

- performing bioaccumulation tests for phytoremediation in the control environment using *C. benghalensis*, *E. pyramidalis* and *P. purpureum* for soils polluted by heavy metals to evaluate their performances;
- assessing the bioaccumulation capacities of *Ipomoea aquatica*, *Panicum maximum*, *Setaria barbata* and others species identified in this study that can be used and implemented in the phytoremediation of metal polluted areas in sub-saharian Africa;
- modelling the transfer of heavy metals from polluted lowland soils to macrophyte species in phytoremediation attempts.

IV.3. Recommendations and suggestions

At the end of this study, some recommendations and suggestions were formulated for the attention of the main actors involved in lowlands management chain.

To the stakeholders (population, farmers, contractors, urban councils and researchers):

- Sensitize, inform and train all the stakeholders for better knowledge on the heavy metals pollution transfer and the human risks;
- Put a safety belt using selected remediation plants around the lowlands for their development.

To the Cameroon government:

- strengthen regulations must be exercised by the Cameroonian government on the lowland sites and river exploitation;
- provide information, educational resources and a forum for various stakeholders in urban agricultural areas so that to a better knowledge by citizens of soil-plant-air transfer of pollutants and nutrients is essential to rationalise the risks.
- lowlands must be integrated into the CDP (Communal Development Plan) of the city of Yaounde to preserve agricultural uses and allow food control and sanitation.

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APPENDIX

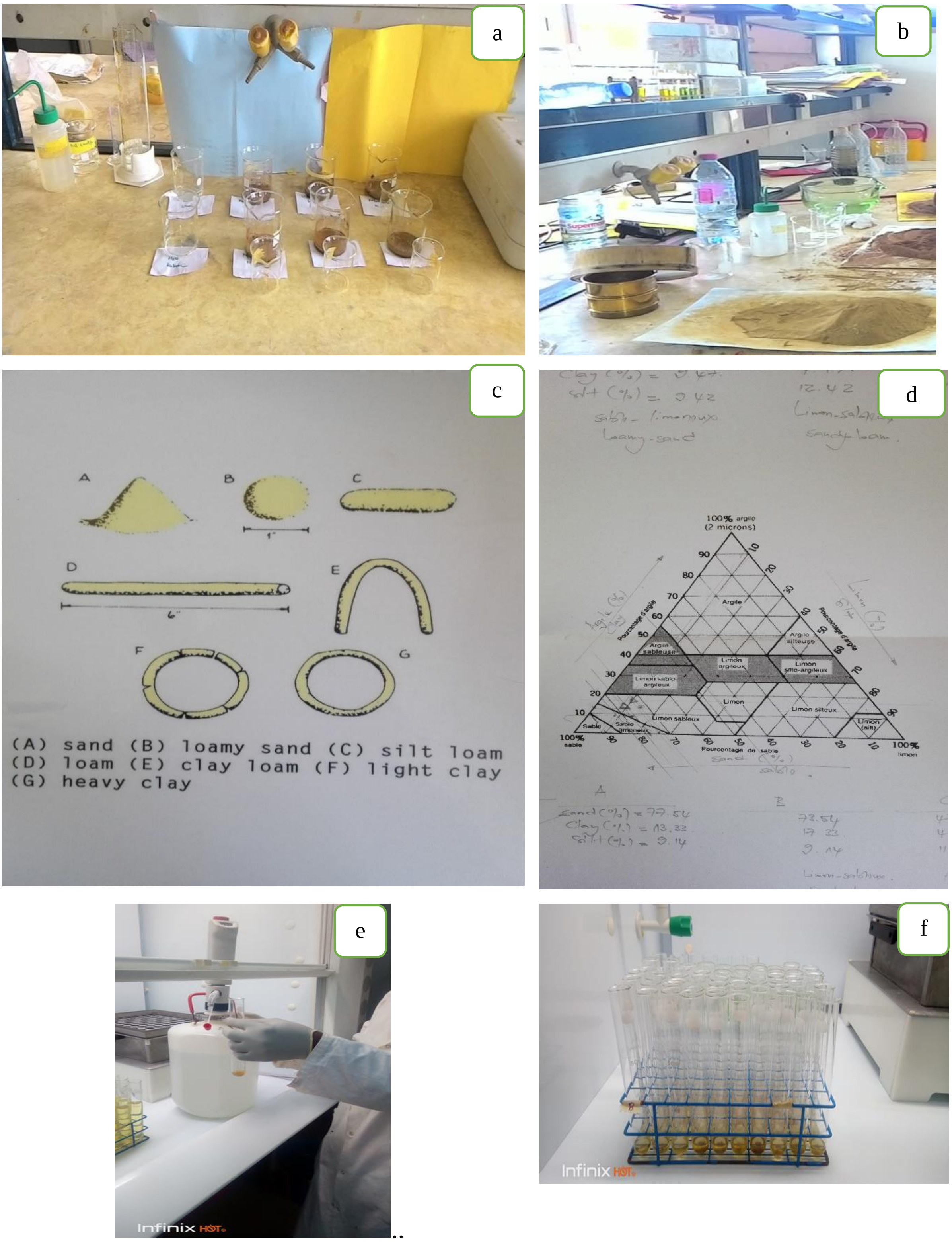
Appendix 1. Quadrats method used for floristic survey in different lowlands.


Appendix 2. Field sampling in lowlands



Appendix 3. Some of the laboratory apparatus used for this study. (a: GPS Map 64s Garmin; b: Auger round-tipped, iron support, handle wrench vt-66 chrom-vanadium, trowel santos 22cm, meter Stanley tylon 5m; c: Calibration solution; d: Multi 3320 meter box WTW; e & f: ICP-OES Spectrometer PerkinElmer - Model Optima™ 8300).

Appendix 4. View of the laboratory during the manipulation of the physico-chemical analysis of soils, determination of soils texture and heavy metals analysis. (a: Preparation of soil samples for measurements of physico-chimical measures; b: Air-dried, crushed and sieved soils in the laboratory; c: Key to the soil textural classes; d: Soil classifications plotted on the USDA texture Triangle; e & f: Digested and diluted water, soil and plants for ICP determination).



Appendix 5. List of macrophytes species identified as bioaccumulators in the 12 lowlands sites of Yaounde in function of their rank/proportion of abundance-dominance.

Macrophytes species	**Rank**	**Abundance**	**Proportion (%)**
Echinochloa pyramidalis	1	2874	12.3
Ipomoea aquatica	2	1984	8.5
Commelina benghalensis	3	1911	8.2
Pennisetum purpureum	4	1516	6.5
Setaria barbata	5	974	4.2
Panicum maximum	6	946	4.1
Ipomoea batatas	7	672	2.9
Alternanthera sessilis	8	524	2.2
Alchornea cordifolia	9	472	2.0
Cynodon dactylon	10	464	2.0
Polygonum salicifolium	11	371	1.6
Colocasia esculenta	12	350	1.5
Leersia hexandra	13	302	1.3
Rhynchospora corymbosa	14	299	1.3
Acanthospermum hispidum	15	294	1.3
Zehneria scabra	16	286	1.2
Cyperus papyrus	17	280	1.2
Musa sapienza	18	275	1.2
Ageratum conyzoides	19	270	1.2
Musa bafousiana	20	260	1.1
Ricinus communis	21	249	1.1
Vernonia amygdalina	22	249	1.1
Justicia cornea	23	248	1.1
Alternanthera ficoidea	24	244	1.0
Ipomoea indica	25	243	1.0
Amaranthus spinosus	26	231	1.0
Luffa aegyptiaca	27	227	1.0
Eleusine indica	28	221	0.9
Amaranthus esculentus	29	213	0.9
Lemna minor	30	200	0.9
Tithonia diversifolia	31	200	0.9
Vernonia calvoina	32	200	0.9
Costus afer	33	195	0.8
Saccharum officinarum	34	194	0.8
Justicia sp	35	181	0.8
Amaranthus grassilis	36	169	0.7
Mitragyna stipulosa	37	165	0.7
Xanthosoma mafaffa	38	165	0.7

Acroceras zizanioides	39	148	0.6
Cucumeropsis mannii	40	147	0.6
Triumfetta pentandra	41	142	0.6
Pueraria phaseoloides	42	125	0.5
Telfairia occidentalis	43	125	0.5
Nymphea lotus	44	114	0.5
Lycopersicum esculentum	45	103	0.4
Nymphea alba	46	102	0.4
Bambusa vulgaris	47	100	0.4
Angelonia angustifolia	48	93	0.4
Asystasia gangetica	49	90	0.4
Ipomoea preussii	50	90	0.4
Musa parasidiaca	51	85	0.4
Sida corymbosa	52	85	0.4
Albizia zizia	53	80	0.3
Manihot esculenta	54	80	0.3
Voacanga africana	55	80	0.3
Galensoca ciliata	56	73	0.3
Ludwigia abyssinica	57	70	0.3
Carica papaya	58	69	0.3
Raphia hookeri	59	65	0.3
Chromolaena odorata	60	64	0.3
Cananga odorata	61	60	0.3
Cyperus alternifolius	62	57	0.2
Alternanthera sp	63	55	0.2
Solanum aethiopum	64	54	0.2
Vigna radiata	65	50	0.2
Diplansium sammatii	66	49	0.2
Eclipta prostrata	67	48	0.2
Mariscus flabelliformis	68	46	0.2
Paspalum conjugatum	69	45	0.2
Acmella uliginosa	70	41	0.2
Combretum zenkeri	71	40	0.2
Ludwigia sp	72	40	0.2
Passiflora foetida	73	40	0.2
Pseudospondias microcarpa	74	40	0.2
Senna alata	75	40	0.2
Sida rhombifolia	76	40	0.2
Drymaria cordata	77	39	0.2
Physalis angulata	78	37	0.2
Eucalyptus globulus	79	35	0.2
Phyllanthus amarus	80	34	0.1

Macrophytes species	Rank	Abundance	Proportion (%)
Ludwigia decurrens	81	32	0.1
Laportea ovalifolia	82	31	0.1
Zea mays	83	31	0.1
Ipomoea sp	84	30	0.1
Marantochloa purpurea	85	30	0.1
Strelitzia regina	86	30	0.1
Oplismenus burmannii	87	28	0.1
Polygonum lanigerum	88	28	0.1
Laportea aetuans	89	27	0.1
Dacryodes edulis	90	25	0.1
Mimosa invisa	91	25	0.1
Paullinia pinnata	92	25	0.1
Triumfetta cordifolia	93	25	0.1
Hewittia sublobata	94	23	0.1
Xanthosoma pubescens	95	22	0.1
Centella asiatica	96	21	0.1
Cyperus sp	97	21	0.1
Cyathula prostrata	98	20	0.1
Hibiscus syriacus	99	20	0.1
Indigofera sp	100	20	0.1
Mangifera foetida	101	20	0.1
Crotalaria sp	102	20	0.1
Canna indica	103	19	0.1
Emilia praetermissa	104	19	0.1
Mimosa pudica	105	17	0.1
Solanum nigrum	106	17	0.1
Cucumis melo	107	15	0.1
Cyperus esculentus	108	15	0.1
Dioscorea bulbifera	109	15	0.1
Ficus citrifolia	110	15	0.1
Abelmoschus esculentus	111	14	0.1
Cleome ciliata	112	14	0.1
Phaseolus vulgaris	113	14	0.1
Solanum torvum	114	14	0.1
Luffa sp	115	13	0.1
Oxalis corniculata	116	13	0.1
Axonopus compressus	117	12	0.1
Citrullus lanatus	118	12	0.1

Macrophytes species	Rank	Abundance	Proportion (%)
Lactuca taraxacifolia	119	11	<0.1
Ludwigia hyssopifolia	120	11	<0.1
Abrus precatorius	121	10	<0.1
Acalypha hispida	122	10	<0.1
Ceiba pentandra	123	10	<0.1
Corchorus olitorius	124	10	<0.1
Cymbopogon citratus	125	10	<0.1
Dioscorea sp	126	10	<0.1
Dioscorea dumetorum	127	10	<0.1
Elaeis guineensis	128	10	<0.1
Euphorbia heterophylla	129	10	<0.1
Nicotiana tabaccum	130	10	<0.1
Pistia stratiotes	131	10	<0.1
Psidium guajava	132	10	<0.1
Pterygota sp	133	10	<0.1
Vernonia sp	134	10	<0.1
Odontonema strictum	135	8	<0.1
Milicia excelsa	136	7	<0.1
Ricinodendron heudelotii	137	7	<0.1
Bidens pilosa	138	6	<0.1
Capsicum frutescens	139	6	<0.1
Ficus exasperata	140	6	<0.1
Persea americana	141	6	<0.1
Scoparia dulcis	142	6	<0.1
Sida acuta	143	6	<0.1
Centrosema pubescens	144	5	<0.1
Cleome rudidosperma	145	5	<0.1
Clerodendrum buchholzi	146	5	<0.1
Dissotis rotundifolia	147	5	<0.1
Eragrostis tenella	148	5	<0.1
Eremomastax speciosa	149	5	<0.1
Erigeron floribundus	150	5	<0.1
Euphorbia hirta	151	5	<0.1
Ficus umbellata	152	5	<0.1
Ipomoea involucrata	153	5	<0.1
Ipomoea mauritiana	154	5	<0.1
Moringa oleifera	155	5	<0.1
Oplismenus hirtellus	156	5	<0.1
Paspalum sp	157	5	<0.1

Macrophytes species	Rank	Abundance	Proportion (%)
Pteridium aquilinum	158	5	<0.1
Rektophylum mirabile	159	5	<0.1
Solenostemon monostachyus	160	5	<0.1
Sorghum arundinaceum	161	5	<0.1
Acalypha crenata	162	4	<0.1
Cyperus iria	163	4	<0.1
Oxalis barbieri	164	4	<0.1
Vernonia cinerea	165	4	<0.1
Datura species	166	3	<0.1
Digitaria horizontalis	167	3	<0.1
Mirabilis jalapa	168	3	<0.1
Piper umbellatum	169	3	<0.1
Roystonea regia	170	3	<0.1
Senna occidentalis	171	3	<0.1
Solanum macrocarpum	172	3	<0.1
Synedrella nodiflora	173	3	<0.1
Theobroma cacao	174	3	<0.1
Cayratia debilis	175	2	<0.1
Corchorus ditorus	176	2	<0.1
Cyperus longibracteatus	177	2	<0.1
Dichrocephala integrifolia	178	2	<0.1
Duranta erecta	179	2	<0.1
Emilia coccinea	180	2	<0.1
Ipomoea religiosa	181	2	<0.1
Mucuna pruriens	182	2	<0.1
Phyllanthus sp	183	2	<0.1
Portulaca oleracea	184	2	<0.1
Spinacia oleracea	185	2	<0.1
Sporobolus pyramidalis	186	2	<0.1
Trema orientalis	187	2	<0.1
Vigna unguiculata	188	2	<0.1
Kyllinga bulbosa	189	1	<0.1

Appendix 6. Correlation values between heavy metals and the physico-chemical properties of soils and their significance.

	pH_Kcl	PH_H$_{2}$0	EC	TDS	Sal	T	CEC	OC	OM	Sand	Clay	Silt	Pb	Cu	Cr	As	Ni	Cd	Co	Zn
pH_Kcl	1																			
PH_H20	0.732**	1																		
EC	-0.210	-0.580*	1																	
TDS	-0.297	-0.642**	0.989**	1																
Sal	-0.099	-0.075	0.617**	0.616**	1															
T	-0.361	-0.735**	0.895**	0.901**	0.407	1														
CEC	0.095	0.023	0.270	0.238	0.275	0.083	1													
OC	0.151	0.012	0.352	0.319	0.314	0.145	0.991**	1												
OM	0.093	-0.076	0.449	0.420	0.363	0.247	0.977**	0.994**	1											
Sand	0.315	0.139	0.087	0.111	0.116	0.052	-0.708**	-0.613**	-0.590*	1										
Clay	-0.317	-0.070	-0.261	-0.287	-0.243	-0.180	0.558*	0.447	0.411	-0.976**	1									

Silt	-0.191	-0.310	0.508*	0.495*	0.338	0.378	0.933**	0.929**	0.953**	-0.710**	0.540*	1								
Pb	-0.266	-0.228	0.165	0.281	0.286	0.107	-0.309	-0.248	-0.210	0.592**	-0.643**	-0.210	1							
Cu	0.513*	0.591**	-0.130	-0.123	0.428	-0.355	-0.205	-0.147	-0.175	0.671**	-0.678**	-0.400	0.492*	1						
Cr	0.109	-0.092	0.313	0.297	0.109	0.163	0.946**	0.960**	0.956**	-0.611**	0.448	0.913**	-0.198	-0.282	1					
As	-0.338	-0.609**	0.891**	0.911**	0.573*	0.773**	0.574*	0.630**	0.711**	-0.216	0.015	0.788**	0.216	-0.234	0.625**	1				
Ni	0.136	-0.043	0.375	0.325	0.232	0.211	0.975**	0.977**	0.975**	-0.688**	0.533*	0.937**	-0.416	-0.307	0.946**	0.615**	1			
Cd	-0.187	-0.499*	0.895**	0.881**	0.572*	0.756**	0.662**	0.716**	0.788**	-0.290	0.089	0.835**	-0.004	-0.257	0.681**	0.968**	0.732**	1		
Co	-0.076	-0.013	0.125	0.075	0.154	0.033	0.904**	0.846**	0.826**	-0.915**	0.828**	0.860**	-0.583*	-0.445	0.781**	0.399	0.894**	0.518*	1	
Zn	0.400	0.207	0.359	0.316	0.420	0.081	0.899**	0.940**	0.928**	-0.339	0.166	0.776**	-0.136	0.166	0.866**	0.558*	0.882**	0.655**	0.664**	1

**. La corrélation est significative au niveau 0.01 (bilatéral).
*. La corrélation est significative au niveau 0.05 (bilatéral).

Appendix 7. Correlation coefficient of physicochemical and heavy metals in lowland water and their significance (both rainy and dry seasons).

	Rainy season											
	pH	EC	TDS	T	Eh	Sal	Pb	Cu	As	Ni	Co	Zn
pH	1											
EC	0.314	1										
TDS	0.320	1.000**	1									
T	0.042	.883**	.881**	1								
Eh	-0.571	-.934**	-.940**	-.771*	1							
Sal	-0.470	0.449	0.454	0.563	-0.334	1						
Pb	-.707*	-0.057	-0.051	0.125	0.154	.866**	1					
Cu	-0.369	-0.432	-0.424	-0.505	0.430	0.000	0.250	1				
As	-.707*	-0.057	-0.051	0.125	0.154	.866**	1.000**	0.250	1			
Ni	-0.600	0.222	0.227	0.372	-0.112	.971**	.961**	0.120	.961**	1		
Co	-.707*	-0.057	-0.051	0.125	0.154	.866**	1.000**	0.250	1.000**	.961**	1	
Zn	-0.051	.873**	.876**	.873**	-.775*	.826**	0.434	-0.282	0.434	.666*	0.434	1

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

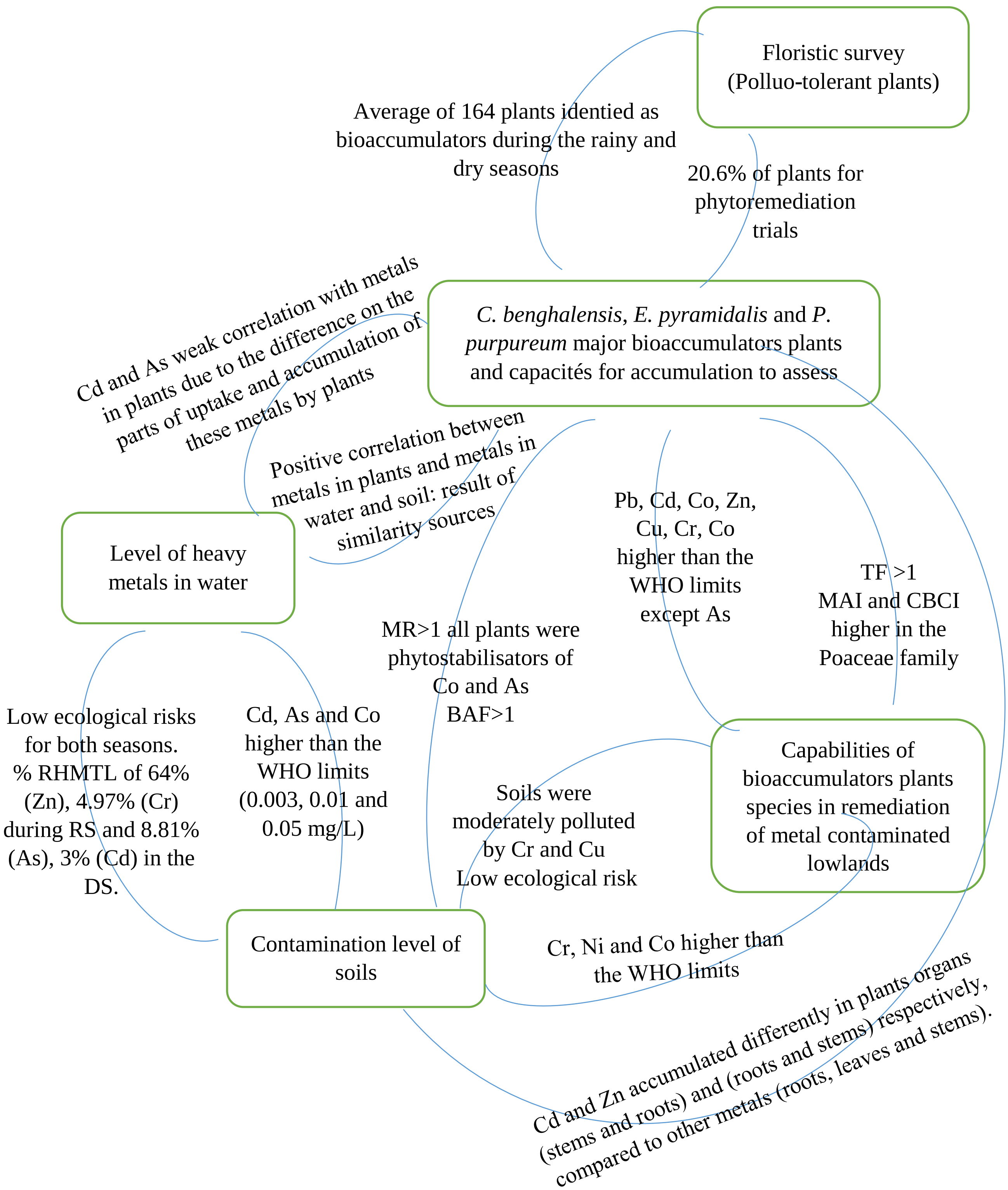
	Dry season													
	pH	EC	TDS	T	Eh	Sal	Pb	Cu	Cr	As	Ni	Cd	Co	Zn
pH	1													
EC	.711*	1												
TDS	.733*	.990**	1											
T	0.491	.669*	0.658	1										
Eh	-0.351	-.796*	-.782*	-.747*	1									
Sal	-0.089	0.204	0.203	0.108	0.079	1								
Pb	-0.584	-.772*	-.749*	-0.584	.831**	0.428	1							
Cu	0.475	0.620	0.592	0.451	-.740*	-0.592	-.975**	1						
Cr	0.116	0.187	0.219	0.213	0.091	.901**	0.453	-0.638	1					
As	-0.178	-.685*	-.687*	-0.230	0.415	-.707*	0.216	-0.062	-0.503	1				
Ni	0.536	.705*	.679*	0.525	-.793*	-0.508	-.995**	.993**	-0.543	-0.145	1			
Cd	-0.177	-.684*	-.686*	-0.229	0.414	-.707*	0.216	-0.062	-0.503	1.000**	-0.144	1		
Co	-0.178	-.685*	-.686*	-0.229	0.415	-.707*	0.216	-0.062	-0.503	1.000**	-0.144	1.000**	1	
Zn	0.179	0.213	0.179	0.110	-0.438	-.835**	-.772*	.893**	-.917**	0.262	.834**	0.262	0.262	1

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

Appendix 8. Pictures of investigated lowlands (a. Mvan, b. Mokolo-elobie, c. Biyem-assi, d. Nkol-nso'o (New road Nkolbisson), e. Ngousso, f. IRAD-Nkolbisson, g. Municipal lake, h. Atemengue pond Obili, i. WWTP Cité-verte, j. WWTP Messa, k. Retenue pond, l. Ongot village).


g
h
i
i
k
l

Appendix 8. Functional diagram of the results obtained



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