Magnesium ferrite-nitrogen–doped graphene oxide nanocomposite: effective adsorptive removal of lead(II) and arsenic(III)

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Abstract
Magnetic nanocomposites have received immense interest as adsorbents for water decontamination. This paper presents adsorptive properties of nitrogen-doped graphene oxide (N-GO) with magnesium ferrite (MgFe2O4) magnetic nanocomposite for removing lead(II) (Pb(II)) and arsenite As(III) ions. Transmission electron microscope (TEM) image of synthesized nanocomposite revealed the wrinkled sheets of N-GO containing MgFe2O4 nanoparticles (NPs) with particle size of 5–15 nm distributed over its surface. This nanocomposite displayed higher BET surface area (72.2 m²g⁻¹) than that of pristine MgFe2O4 NPs (38.4 m²g⁻¹). Adsorption on the nanocomposite could be described by the Langmuir isotherm with the maximum adsorption capacities were 930 mg/g, and 64.1 mg/g for Pb(II) and As(III), respectively. Whereas, maximum removal efficiencies were observed to be 99.7 ± 0.2% and 93.5 ± 0.1% for Pb(II) and As(III), respectively. The study on the effect of coexisting anions on the adsorption of metal ions showed that the phosphate ions were potential competitors of Pb(II) and As(III) ions to adsorb on the nanocomposite. Significantly, the investigation on adsorption of metal ion in the presence of coexisting heavy metal ions indicated the preferential adsorption of Pb(II) ions as compared to Cd(II), Zn(II) and Ni(II) ions. The effectiveness of the nanocomposite to remove the metal ions in electroplating wastewater was demonstrated.

Keywords Magnesium ferrite · Nitrogen-doped graphene oxide · Pb(II) · As(III) · Adsorption

Introduction
The contamination of surface and ground water is a major threat to public health. Heavy metal ions (Pb(II), Ni(II), Zn(II), Cd(II)), and metalloids viz. As(III) and Cr(VI) in the water bodies is considered as one of the most serious problems ( (Balali-Mood et al. 2021)). Based on the guidelines of the World Health Organization (WHO), lead (Pb) and arsenic (As) are categorized among the priority hazardous substances when their concentrations in water exceeds 50 µg/L and 10 µg/L, respectively ((Ghasemabadi et al. 2018); (Shaji et al. 2021)). Contamination of Pb(II) ions mainly arises from the effluent wastes from industries such as mining, galvanization, smelting, and battery manufacturing. The presence of Pb(II) ions in water could increase the health risk of high blood pressure, kidney damage, abdominal cramps and headache ((Zou et al. 2019)). Similarly, arsenic pollution results from a variety of anthropogenic activities such as mining, smelting of ores, and burning of fossil fuels ((Briffa et al. 2020)). Different oxidation states of arsenic exist in the environment such as arsenide (As3−), arsenite (AsO3 3−) and arsenate (AsO4 3−) ((Wei et al. 2017)). As(III) is more toxic, soluble, and mobile species than As(V) ((Feng et al. 2018)). Acute toxicity of As(III) is associated with abdominal pain, heart disease and diarrhea. Chronic exposure to As(III) can lead to multisystem failure ((Salih et al. 2019)).

Various remediation techniques such as chemical precipitation, coagulation, dissolved air-floatation, flocculation, adsorption, and biological methods are applied in removing
toxic metal ions in water (Hoang et al. 2020); (Neelaveni et al. 2019). Among different methods, adsorption is considered as an effective method due to its ease of operation (Sharma et al. 2015); (Wang et al. 2021)). In the recent years, carbonaceous nanomaterials have been studied extensively as promising adsorbents for separating heavy metal ions and metalloids from water (Kaur et al. 2021); Nejadshaﬁee et al. 2020). Graphene is a crystalline allotrope of carbon with two-dimensional properties, its functionalized form, i.e., graphene oxide (GO) interacts with adsorbates through hydroxyl, carboxyl, and epoxy functional groups on its surface (Grewal and Kaur 2017); (Kaur et al. 2018)). Significantly, the magnetic nanocomposites have moved to the forefront of the materials research due to their tailored structure, high porosity, magnetic properties, and extraordinary capability to selectively adsorb large quantities of contaminants (Ismail and Jaber 2018). Ferrite nanoparticles (NPs), mixed metal oxides with iron oxide, and carbon-based materials represent magnetic nanocomposites and have been considered as efﬁcient adsorbents for the removal of heavy metals and metalloids (Kavita et al. 2018); (Chu et al. 2019); (Arul et al. 2017)). Interactions responsible for adsorption behavior include hydrogen bonding, ion-exchange, inner-sphere/outer-sphere complex formation, and dipole–dipole interactions depending on the nature of contaminants, adsorbents, and adsorption conditions (Tran et al. 2020); (Lingamdinne et al. 2019).

Among various ferrite NPs, MgFe2O4 NPs are considered as suitable adsorbents owing to their chemical and thermal stability along with their size uniformity (Khushboo et al. 2021). They are water insoluble, non-toxic and have many surface-active sites which strengthen their use in fabricating nanocomposites (Abraham et al. 2018). In the past few years, the magnetic nanocomposites have been examined for their adsorptive properties (Silambarasu et al. 2018); (Lingamdinne et al. 2020); (Kaur and Kaur 2020). However, nitrogen (N)-doped GO (NGO)-MgFe2O4 nanocomposite has not been explored for its adsorptive properties. The current study was aimed to enhance the adsorptive efficiency of GO-MgFe2O4 nanocomposite by doping with nitrogen (N) and optimizing ratio of NGO and MgFe2O4.

In the present paper, synthesis of nitrogen-doped GO sheets was carried out, which were then nanofabricated with MgFe2O4 NPs to prepare MgFe2O4-NGO nanocomposite. The effect of nitrogen doping GO and fabricating its magnetic nanocomposite on the adsorptive removal of Pb(II) and As(III) was examined in detail. Experiments were performed under various conditions of adsorption dose and pH to gain insight into the adsorptive properties of MgFe2O4-NGO nanocomposite. Studies of adsorption of Pb(II) and As(III) ions were performed to gain insight into the adsorptive properties of MgFe2O4-NGO nanocomposite. Results were compared with the pristine GO, N-GO, and MgFe2O4 NPs under similar experimental conditions. Adsorption isotherms, kinetic modeling, and XPS studies were used to gain a detailed insight on the adsorption mechanism of Pb(II) and As(III) ions onto MgFe2O4-NGO nanocomposite. The effect of competing cations and anions present in water bodies was also ascertained. The studies were extended to electroplating industrial wastewater.

**Materials and methods**

**Synthesis and characterization**

The chemicals used in this study, including graphite powder, ortho-phosphoric acid, sulfuric acid, hydrogen peroxide, ammonium hydroxide, potassium permanganate, hydrochloric acid, sodium chloride, sodium carbonate, sodium phosphate, ferric nitrate, magnesium nitrate, lead nitrate, and sodium arsenite were of analytical reagent grades with high purity and were purchased from Molychem, India. De-ionized water was used to prepare solutions of different concentrations.

The flowchart depicting the synthesis of pristine GO, N-GO, MgFe2O4 NPs, and MgFe2O4-NGO nanocomposite is presented in Scheme S1. GO was prepared using the modified Hummer’s method (Zabba et al. 2017)), which involved the mixing of 1.0 g graphite powder with 90.0 mL of sulfuric acid and 10.0 mL of ortho-phosphoric acid. This solution was kept in ice bath and stirring was carried out for 20.0 min. Subsequently, 4.5 g KMnO4 was added slowly to the reaction mixture and stirred continuously for another 12 h. Then, in a 200-mL beaker, the crushed ice was taken to which 3.0 mL of 30% H2O2 was added. This solution was added to the reaction mixture slowly and the solution was continuously stirred, which resulted in the formation of green precipitates. The precipitates were filtered using the suction filtration and washed with distilled water several times to get the neutral pH. Afterward, the mixture was allowed to dry in an oven at 60 °C. In the synthesis of MgFe2O4, 2.0 mol of Fe(NO3)3.9H2O and 1.0 mol of Mg(NO3)2.6H2O were dissolved in 20 ml de-ionized water. To this aqueous solution, 2.2 mol of citric acid was added. The above solution was magnetically stirred at 60 °C to which NH4OH solution was added to make the pH of solution to 7.0 and the solution transformed into sol. After continuous stirring for 8.0 h, the sol changed into gel, which was dried at 100 °C for 12.0 h and its volume increased by 5 times. Finally, the dried gel was ground and calcined at 300 °C for 3.0 h to get MgFe2O4 as the final product.

The N-GO was synthesized using the ultrasonication method by applying 60 W power and 20 kHz frequency. The pristine GO (0.5 g) was dissolved in 20.0 mL of 25% NH4OH solution at 25 °C, and sonicated for different time
intervals of 15, 30, 60, and 120 min. The prepared solutions were centrifuged at 5000 rpm, followed by repeated washing with water. The obtained black precipitates were dried in an oven at 60 °C overnight to obtain N-GO. Time period of 30 min was determined to be optimum for N-doping (see CHN analysis in Fig. S1).

In preparing MgFe$_2$O$_4$-NGO nanocomposite, N-GO and MgFe$_2$O$_4$ NPs were dissolved separately in different w/w ratio in 15.0 mL ethanol and sonicated for 30 min. The two solutions were mixed and ultrasonicated again for 1.0 h. The resulting mixture was centrifuged, and the obtained brown colored MgFe$_2$O$_4$-NGO nanocomposite was washed with water and dried in an oven at 60 °C overnight.

Different characterization techniques were employed to study functional groups, structure, composition, particle topography, morphology, surface area, surface charge, and magnetic properties along with instrumental details are given in Table S1.

### Adsorption measurements

The stock solutions (1000.0 mg/L) of Pb(II) and As(III) ions were prepared by dissolving the corresponding metal salts in 1.0 L of double-distilled water. Working solutions were prepared by dilution of stock solutions using the deionized water. The effect of pH was studied by adjusting the solution pH with either 0.1 M NaOH or 0.1 M HCl (Li et al. 2020). The optimum pH for aqueous adsorption of Pb(II) and As(III) ions by synthesized adsorbents was measured, while the effect of adsorbent dosage (0.1–3.0 g/L) was also studied under the optimum pH. Adsorption isotherm studies were conducted at 25 °C by varying the concentrations (5.0–100.0 mg/L) of Pb(II) and As(III) ions with an adsorbent dose of 0.01 g and optimized pH of 6.0 and 7.0, respectively. The adsorption data were fitted using Freundlich, Langmuir, Temkin, and Dubinin-Radushkevich (D-R) adsorption isotherms. To analyze the adsorption kinetics, pseudo-first order and pseudo-second order models were applied. The details of the isotherms and kinetic models are presented in Table S2. Also, the best fit, suitability, and agreement of kinetic and isotherm models were validated using the statistical analysis, i.e., sum of squared error (SSE), chi-square error ($\chi^2$), root mean square error (RMSE), ‘F’-statistic and Log Likelihood Error ($G^2$) as discussed in Table S2. To determine the adsorption equilibrium time, the contact time was varied from 0 to 180 min under the optimized conditions of adsorbent dose and pH. The batch adsorption experiments were conducted at 25.0 ± 1 °C and the solutions were shaken in an orbital shaker operated at 130 rpm.

The influence of other divalent heavy metal ions in the quaternary Pb–Cd–Zn–Ni system was studied using respective metal nitrates and the influence of common co-existing anions (Cl$^-$, CO$_3^{2−}$, and PO$_4^{3−}$) using NaCl, Na$_2$CO$_3$, and Na$_2$(PO$_4$)$_3$ on adsorption of Pb(II) and As(III) ions was also investigated. The concentrations of these salts were varied from 0.25 to 2.0 mg/L. The initial concentrations of Pb(II) and As(III) ions were used with an optimized adsorbent dose of MgFe$_2$O$_4$-NGO as 2.0 g/L for both Pb(II) and As(III) ions. The remaining concentrations of Pb(II) and As(III) ions were determined. The removal of metal ions from electroplating wastewater was also carried out using the prepared nanocomposite.

Recyclability tests were performed using 0.20 g of adsorbent in 100.0 mL solution containing 10.0 mg/L and 5.0 mg/L concentrations of Pb(II) and As(III) solutions, respectively. The solutions were shaken for 2.0 h at 130 rpm and the centrifuges were analyzed to measure the concentrations of Pb(II) and As(III) ions. The adsorbent was washed with 0.1 M HCl, centrifuged, and repeatedly washed with deionized water several times. Finally, the adsorbent was collected by centrifugation and dried at 65 °C in an oven for the next adsorption/desorption cycle. This process was repeated for six adsorption–desorption cycles to determine the reusability of the adsorbent.

### Results and discussion

#### Characterization of synthesized adsorbents

In the TEM images, the wrinkled layers of GO sheets were observed (Fig. 1a). This could be ascribed to thermodynamic instability of exfoliated GO sheets and Van der Waal attractions between GO layers (Grewal and Kaur 2017; Abbasi et al. 2021). The internal and external coupling of N-GO sheets was perceived to produce microwrinkling, which could increase the number of layers and length of the layered structure upon nitrogen doping, observed previously in the TEM image of N-GO (Fig. 1b) (Alyaa et al. 2016). MgFe$_2$O$_4$ NPs displayed agglomeration in TEM images which is a characteristic of magnetic NPs with the particle size ranging from 5 to 15 nm (Fig. 1c). Also, the magnetic nature of pristine NPs resulted in some clusters as observed in the TEM image. In case of nanocomposite, wrinkled sheets of N-GO contained the MgFe$_2$O$_4$ NPs distributed over their surface (Fig. 1d). The results also showed the inclusion of N-GO nanosheets prevented the aggregation of MgFe$_2$O$_4$ NPs to form larger aggregates.

SEM images using energy-dispersive X-ray (SEM–EDX) were examined for comparing the surface topography and composition of GO, N-GO, MgFe$_2$O$_4$ NPs, and nanocomposite (Fig. S2). GO displayed a layered structure with crumpled edges with 70.9% C content and 29.1% O content (Fig. S2a). Comparatively, N-GO showed the flaky morphology with slightly wrinkled and folded features. EDX mapping...
(Fig. S2b) suggested the doping of N into GO. The SEM image of MgFe$_2$O$_4$ NPs depicted the agglomeration owing to attractive magnetic forces which decreased the surface energy and EDX mapping displayed the presence of Mg, Fe and O (Fig. S2c). Nanocomposite presented rough structure in SEM image (Fig. S2d). This may be due to deposition of MgFe$_2$O$_4$ on N-GO with EDX mapping depicting the presence of Mg, Fe, N, C, and O, thus confirming the nanofabrication of N-GO with MgFe$_2$O$_4$ NPs.

BET surface area analysis was performed to determine the change in surface area due to the nitrogen doping into GO and its fabrication with MgFe$_2$O$_4$ NPs. The N$_2$ adsorption–desorption isotherms are presented in Fig. S3 along with the pore size distribution (inset), which was predicted by plotting pore volume vs. pore diameter. Synthesized adsorbents followed the type IV isotherm of H3 type hysteresis loop, indicating their slit-shaped mesoporous nature. As listed in Table S3, the BET surface area and pore volume of GO were observed to be 81.4 m$^2$/g$^{-1}$ and 5.0 x 10$^{-3}$ cm$^3$/g$^{-1}$, respectively. Comparatively, these parameters for N-GO were 178.2 m$^2$/g$^{-1}$ and 1.3 x 10$^{-3}$ cm$^3$/g$^{-1}$, which were significantly higher than that of GO. Nitrogen doping on its surface caused the additional exfoliation and perforation of sheets. Therefore, the increase in surface area was due to more wrinkled sheets in N-GO as compared to GO (Wang et al. 2020). However, the BET surface area and pore volume for MgFe$_2$O$_4$ NPs and nanocomposite were 38.39 m$^2$/g$^{-1}$ and 72.20 m$^2$/g$^{-1}$ and 0.039 cm$^3$/g$^{-1}$ and 0.053.
cm\(^3\)g\(^{-1}\), respectively. The higher surface area for nanocomposite as compared to MgFe\(_2\)O\(_4\) NPs was attributed to the effective surface modulation in the presence of N-GO. Also, the increased mesoporosity was observed for nanocomposite as compared to pristine MgFe\(_2\)O\(_4\) NPs due to presence of N-GO.

Effect of change in applied magnetic field (H, Oe) on magnetization or magnetic flux (M, emu\(^g\)\(^{-1}\)) of the synthesized NPs and nanocomposite is given in Fig. S4. They displayed s-shaped narrow hysteresis loop signifying ferromagnetic behavior at room temperature. The magnetic parameters of the adsorbents are reported in Table S4. The Ms value for MgFe\(_2\)O\(_4\) NPs and nanocomposite was 8.55 and 2.74 emu\(^g\)\(^{-1}\), respectively. The Ms value of nanocomposite decreased on addition of non-magnetic N-GO which may be correlated to the quenching of magnetic momentum in them. The magnetic properties are advantageous for its magnetic separation from aqueous medium.

X-ray diffraction patterns of GO showed the distinctive peaks at 2θ = 10.88˚ (hkl = 001) and 42.41˚ (hkl = 100), confirming the conversion of graphite into GO (Fig. 2). A similar XRD pattern was observed for N-GO with a shift of the peak corresponding to 001 plane towards a smaller 2θ value of 9.88°. This may be due to the expansion of interlayer distance by N insertion onto the GO surface (Tao et al. 2017). MgFe\(_2\)O\(_4\) NPs displayed the distinctive peaks at 2θ = 29.9˚ (hkl = 220), 35.0˚ (hkl = 311), 42.5˚ (hkl = 400), 56.5˚ (hkl = 511), and 62.1˚ (hkl = 440), confirming the presence of crystalline cubical spinel structure (Biehl et al. 2018). All the observed peaks and corresponding Miller indices (hkl) of the fcc lattice were verified by matching with ASTM Data Card No. 17–465. Additionally, the average particle size, calculated from the XRD parameters by using Debye–Scherrer equation, was 13.5 nm, which agreed well with the particle size obtained from TEM studies (Fig. 1c). Whereas, in case of nanocomposite in addition to all the above phases identified for pristine NPs, an additional peak at 2θ = 9.84˚ corresponding to (001) plane was assigned to the presence of N-GO.

The FT-IR spectrum of GO displayed the presence of different functional groups, i.e., -C\(=\)O, -O–H, and –COOH (Fig. S5). The absorption bands centered at 3400 cm\(^{-1}\) and 2900 cm\(^{-1}\) were assigned to the stretching vibrations of -O–H and -C-H, respectively. The other bands observed at 1739 cm\(^{-1}\), 1560 cm\(^{-1}\), and 1228 cm\(^{-1}\), corresponded to the stretching vibrations of >C=O in carboxyl and carbonyl groups, -C= C vibrations, and C-O stretching vibrations, respectively (Ranjan et al. 2018). The FT-IR spectrum of N-GO indicated the absorption bands at 3400 cm\(^{-1}\), 3006 cm\(^{-1}\), 1738 cm\(^{-1}\), and 1560 cm\(^{-1}\), which individually corresponded to the stretching vibrations of –O–H, -N–H, >C =O, and -C= C, respectively. Significantly, two new bands at 1399 cm\(^{-1}\) and 1243 cm\(^{-1}\) appeared in FT-IR spectrum of N-GO, which belong to the stretching vibrations of -C=N and -C-N, respectively (Tao et al. 2017). MgFe\(_2\)O\(_4\) NPs displayed an absorption band at 3433 cm\(^{-1}\) due to O–H stretching and broad bands centered at 572, 468 and 440 cm\(^{-1}\) corresponding to the M–O bond vibrations (Tang et al. 2013). Similarly, FT-IR spectrum of MgFe\(_2\)O\(_4\)-NGO nanocomposite indicated the absorption bands at 3153 cm\(^{-1}\) and 1558 cm\(^{-1}\), which corresponded to -O–H/N–H bond stretching and bending vibrations, respectively. Bands at 1387 cm\(^{-1}\) and 1199 cm\(^{-1}\) were attributed to the –C=N and –C=N stretching vibrations along with the characteristic bands for pristine MgFe\(_2\)O\(_4\) NPs at 642 cm\(^{-1}\) and 485 cm\(^{-1}\). O–H stretching and M–O vibrations for MgFe\(_2\)O\(_4\) NPs were observed at 3433 cm\(^{-1}\) and 572 cm\(^{-1}\), respectively. These vibrations shifted to 3153 cm\(^{-1}\) and 642 cm\(^{-1}\) in nanocomposite, respectively. This suggests that the MgFe\(_2\)O\(_4\) NPs were bound to the surface of N-GO via coordinate bond or electrostatic interactions between ferrite NPs and different functionalities of N-GO nanosheets (Huong et al. 2018).

**Adsorption study**

Initially, the percentage removal of Pb(II) (10.0 mg/L) and As(III) (5.0 mg/L) ions from aqueous solution at optimized
pH of 6.0 and 7.0, respectively. was determined using nano-composite with varied ratios of MgFe$_2$O$_4$ and N-GO, i.e., 1:1, 1:2, and 2:1 (Fig. S6). Nanocomposite with 1:2 w/w ratio displayed maximum percentage removal. This could be explained based on magnetic nature of nanocomposites. Nanocomposite in 2:1 ratio exhibited maximum magnetic characteristics, which led to the increase in NP agglomera-tion and resulted in the lowering of surface area required for adsorption. Whereas nanocomposite in 1:2 ratio had less magnetic properties due to greater amount of N-GO in it which significantly lowered the agglomeration of the NPs and provided greater surface area for adsorption as compared to nanocomposites in 1:1 and 2:1 ratio. The results signif-ify the role of N-GO in increasing the adsorption potential of nanocomposite. Thus, nanocomposite with MgFe$_2$O$_4$ and N-GO in 1:2 ratio was used for further adsorption studies.

**Effect of contact time and adsorption kinetics**

The adsorption of Pb(II) and As(III) on the synthesized adsorbents with time was determined (Figs. S7a and S7b). The adsorption on MgFe$_2$O$_4$-NGO nanocomposite was rapid up to 40 min for Pb(II) and As(III) ions and the equilibrium was attained within 60 min. Nanocomposite showed the removal of 94.1 ± 0.5% and 93.4 ± 1.1% in 120 min for Pb(II) and As(III) ions, respectively. In further adsorption studies, equilibration time of 120 min was chosen.

To study the kinetics and the involved mechanism in the adsorption process pseudo-first (Figs. 3a and 4a) and pseudo-second order (Figs. 3b and 4b) models were studied. The values for equilibrium adsorption capacity ($q_e$) and rate constants calculated from pseudo-first order and pseudo-second order models are presented in Table 1. $R^2$ for pseudo-first-order kinetic model ranged from 0.76 to 0.98. The difference in experimental and calculated values of $q_e$ was large when pseudo-first order model was applied (Table 1). The pseudo-second-order model gave higher values of regression coefficients ($>0.99$) with much lower difference between experimental and calculated $q_e$ values. The fitting quality of this model to the experimental data was determined from the values of ‘F’-statistics, sum of squared error, and Chi-square error ($\chi^2$). The values of these parameters are summarized in Table 1. For the model to be significant, the values of SSE, RMSE and $\chi^2$ should be small, whereas the values of ‘F’-statistics should be large. The ‘F’-statistics values were higher for the pseudo-second order model (695 to 6445) than that for pseudo-first order model (57 to 245), which again indicated the better fitting of the former model. The smaller values of SSE ranging from 1.46 to 8.66, $\chi^2$ ranging from 0.01 to 1.05 and RMSE ranging from 0.120 to 0.518 for pseudo-second order kinetic model further con-firmed its better fitting than pseudo-first order model. This supports that both the adsorbate and adsorbent were involved in the rate determination of adsorption. Non-linear modeling of pseudo-first order and pseudo-second order models are presented in supplementary data (Figs. S8 and S9). The difference in experimental and calculated values of $q_e$ was larger for non-linear models as compared to linear models. The greater $\chi^2$ error, SSE and RMSE for non-linear models as compared to linearized models further indicated the better fit of later in the present study.

**Effect of adsorbent dose**

Further, the effect of adsorbent dose on adsorption efficiencies of Pb(II) and As(III) ions was monitored by varying the dosage from 0.1 to 3.0 g/L (Fig. 5a and b). The results showed that the metal ion adsorption increased with an increase in adsorbent dose due to the increased adsorp-tion sites. The nanocomposite exhibited removal efficiency of 91.5 ± 0.3% for Pb(II) ions at 0.10 g/L adsorbent dose which further increased on increasing the adsorbent dose up to 1.0 g/L with the adsorption of 99.7 ± 0.2%. A similar trend was observed for GO, N-GO and MgFe$_2$O$_4$ NPs with lesser removal efficiency as compared to nanocomposite. In case of As(III) ions, nanocomposite showed the removal of...
46.0 ± 0.6% at 0.10 g/L adsorbent dose which increased with increase in adsorbent dose up to 2.0 g/L with the removal efficiency of 93.5 ± 0.1%, respectively, followed by no further increase in removal. This can be correlated with the attainment of adsorption–desorption equilibrium on a further increase in adsorbent dose (Kaur et al. 2019; Wang et al. 2020)). Thus, the optimum dosage for adsorbents was found to be 1.0 g/L and 2.0 g/L for both Pb(II) and As(III) ions, respectively.

Although N-GO had higher surface area than GO, MgFe₂O₄ NPs, and nanocomposite, yet the adsorbents behaved differently in aqueous solutions. The N-GO sheets tend to restack that may have reduced the availability of functional groups for adsorbing metal ions in comparison with pristine MgFe₂O₄ NPs and nanocomposite, which have higher porosity (Table S3). There may also be competition among functional groups for adsorbing the same metal ion to cause the wrapping of N-GO sheets and decreased available surface area for adsorption (Kaur et al. 2019)). The highly porous MgFe₂O₄ NPs, which contained large number of hydroxyl groups on its surface, resulted in higher adsorption capacity due to the availability of mesopores and external surfaces for adsorption. MgFe₂O₄-NGO nanocomposite combined features of both N-GO and MgFe₂O₄ NPs, and provided more porous surface for adsorption and exhibited maximum removal efficiency as compared to GO, N-GO and MgFe₂O₄ NPs.

**Adsorption isotherm study**

Adsorption isotherm studies were performed to elucidate the underlying mechanism for the removal of Pb(II) and As(III) ions by synthesized adsorbents. The adsorption data was fitted in Langmuir, Freundlich, Dubinin-Redushkevitch, and Temkin adsorption isotherms. For Langmuir adsorption isotherm, a graph was plotted between 1/Cₑ vs. 1/qₑ, which yielded linear plots for the synthesized adsorbents (Fig. 6a and b). The linear relationships suggest monolayer adsorption behavior of these ions onto synthesized adsorbents. The calculated values of Langmuir isotherm parameters are presented in Table 2. Values of maximum adsorption capacity ‘qₘ’ ranged between 500 and 930 mg/g for synthesized adsorbents. Whereas, the values of Langmuir constant ‘b’ ranged from 0.27 × 10⁻³—14.0 × 10⁻³ Lmg⁻¹ for Pb(II) adsorption. In the case of As(III) ions, qₘ and ‘b’ values were found to be lower than Pb(II) ions (Table 2), suggesting the higher adsorption affinity of Pb(II) ions onto synthesized adsorbents. The higher adsorption capacity for MgFe₂O₄-NGO nanocomposite was attributed to the larger number of surface-active sites available for adsorption. A summary of the ‘qₘ’ values of Pb(II) and As(III) ions for different adsorbents reported in literature are presented in Table 3. A comparison of results clearly demonstrate that the MgFe₂O₄-NGO nanocomposite synthesized in present study has higher ‘qₘ’ value for Pb(II) ions. Whereas, the ‘qₘ’ value for As(III) ions in present study was higher than Fe-Al hydroxides and Fe–Mn minerals reported in literature. The advantage of nanocomposite is that the MgFe₂O₄ NPs used in its synthesis has Mg(II) as divalent ion, thus is safe to use as compared to other adsorbents containing Mn(II) and Al(II) ions. Thus, this magnetic nanocomposite can be used as a promising adsorbent for removal of Pb(II) and As(III) ions from water. Freundlich (Fig. S10), Dubinin-Redushkevitich (D-R), and Temkin isotherm models, are reported in Text S1 and adsorption parameters presented in Table S5 and Table S6.

Non-linear modeling was also performed for the Langmuir and Freundlich isotherms, which showed the best fit in the present study. The plots were linear for Pb(II) and As(III) using MgFe₂O₄-NGO nanocomposite (Fig. S11). Comparatively, the non-linear parameters for pristine materials and nanocomposite are presented in Table S7. This linearity was due to higher adsorption affinity of the adsorbents in the concentration range of studies (Kaur and Kaur 2020)). This has also been reported that the non-linear behavior was more...
common in natural adsorbents and the response curve of column experiments in non-linear models depended upon the concentration (Burglsser et al. 1993). However, in the present study, the batch experiments indicated better fit of linear models. Also, the greater errors in values of \( \chi^2 \) and \( G^2 \) in non-linear models than the linearized models further indicated the better fit of later.

**Effect of pH on adsorption**

Solution pH plays an important role in adsorptive removal of contaminants as it affects the aqueous chemistry along with the surface bonding sites of the adsorbent. Therefore, it is important to optimize the pH, at which the maximum removal of the contaminant ions can be attained. The effect of initial solution pH on the adsorption of Pb(II) and As(III) ions from aqueous solution was studied in the pH range of 2.0–6.0 for Pb(II) ions and 2.0–10.0 for As(III) ions. In this study, the adsorption capacity of GO, N-GO, MgFeO₄ NPs and nanocomposite for Pb(II) ions increased with the increase in pH (Fig. 7a). In pure water, lead is mainly present as Pb(II) at pH < 7.0. However, with increasing pH, the species such as PbO₂⁺, Pb(OH)⁺, and Pb(OH)₂⁺ dominate over Pb(II) ions. At pH 2.0, the higher concentration of H⁺ was competitive with Pb(II) ions and resulted in the lowered adsorption of Pb(II) ions (Lingamdinne et al. 2017). At pH 6.0, the increased removal of Pb(II) ions may be ascribed to lesser competition between Pb(II) and H⁺ for adsorption sites. When solution pH was higher than 6.0, the surface of the adsorbent exhibited less positive charge and caused an increase in adsorption of Pb(II) ions. Also, the precipitation was observed at pH > 6.0 due to the formation of Pb(OH)₂ (Sreejalekshmi et al. 2009). This was further supported by the ionic product of Pb(OH)₂(1.20×10⁻¹⁸ at pH 7.0) which was greater than the Ksp of 1.43×10⁻²⁰, thus causing precipitation. Thus, to avoid the precipitation adsorption experiments for Pb(II) ions were conducted at pH 6.0.

In the case of As(III) ions, the synthesized adsorbents displayed increased adsorption capacity with increase in pH up to 7.0 (Fig. 7b). At pH 2.0, MgFeO₄-NO₃nanocomposite had removal efficiency of 53.2 ± 0.02%, whereas at pH 7.0 it was 91.2 ± 0.2%. The removal efficiency of nanocomposite decreased to 83.1 ± 0.5% at pH10.0. The maximum adsorption of As(III) ions occurred at pH 7.0. On further increase in pH, the preferential adsorption of OH⁻ ions decreased the adsorption of As(III) ions. Arsenite exists as the As(OH)₃⁻ species in the pH range of 2.0–8.0 (Feng et al. 2012; Sharma et al. 2007). With increase in pH, the formation

| Table 1 Rate constant, equilibrium adsorption capacity, and statistical data of pseudo-first and second-order non-linear kinetic models for the adsorption of Pb(II) and As(III) ions |
|---|
| Kinetic models | Pb(II) | As(III) |
| Pseudo-first order | k₁ (min⁻¹) | qₑ, cal (mg/g) | qₑ, exp (mg/g) | R² | SSE | χ² | RMSE | F |
| GO | 0.05 ± 0.9 | 127.9 ± 0.2 | 148.2 ± 1.2 | 0.95 | 321.41 | 4.12 | 0.640 | 98 |
| N-GO | 0.05 ± 0.1 | 137.2 ± 0.3 | 148.2 ± 0.8 | 0.92 | 517.0 | 0.54 | 0.576 | 101 |
| MgFe₂O₄ | 0.05 ± 0.4 | 176.0 ± 0.3 | 206.9 ± 0.2 | 0.90 | 257.1 | 4.38 | 0.498 | 50 |
| MgFe₂O₄-NGO | 0.05 ± 0.02 | 140.2 ± 0.1 | 228.7 ± 0.3 | 0.90 | 57.82 | 3.46 | 0.405 | 96 |
| Pseudo-second order | k₂ × 10² (g/mg/min) | qₑ, cal (mg/g) | qₑ, exp (mg/g) | R² | SSE | χ² | RMSE | F |
| GO | 0.04 ± 0.1 | 139.2 ± 0.3 | 148.2 ± 1.2 | 0.98 | 5.01 | 2.43 | 0.270 | 822 |
| N-GO | 0.02 ± 0.5 | 142.0 ± 0.2 | 148.2 ± 0.8 | 0.97 | 3.76 | 0.17 | 0.285 | 3214 |
| MgFe₂O₄ | 0.03 ± 0.2 | 215.0 ± 0.6 | 206.8 ± 0.1 | 0.95 | 8.21 | 0.89 | 0.178 | 436 |
| MgFe₂O₄-NGO | 0.0002 ± 0.5 | 214.5 ± 0.01 | 228.7 ± 0.1 | 0.98 | 55.2 | 1.43 | 0.135 | 1503 |

\( k₁ \)-rate constant of pseudo-first order reaction; \( k₂ \)-rate constant of pseudo-second order reaction; \( qₑ, \text{cal} \)-calculated value of adsorption capacity; \( qₑ, \text{exp} \)-experimental value of adsorption capacity; \( R² \)-regression coefficient; \( \chi² \)-sum of squared error; \( RMSE \)-root mean square error; \( F \)-F statistics
of negatively charged arsenic species increases and various anions including AsO(OH)$_2$\(^{-}\), AsO$_2$(OH)$^{2-}$ and AsO$_3$$^{3-}$ are formed (Jekel and Amy 2006). The increased adsorption of As(III) ions on increasing pH up to 7.0 and decrease at pH $> 7.0$ suggested that the electrostatic factors alone were not responsible for the adsorption process (Chandra et al. 2010). Such a trend was also noted during the adsorption of As(III) ions on cellulose loaded with iron oxy-hydroxide (Guo and Chen 2005).

The zeta potential measurements were also performed to characterize the surface charge of the synthesized adsorbents. Generally, the particles having zeta potentials greater than +30 mV or less than -30 mV are considered stable due to electrostatic repulsion (Konkena and Vasudevan 2012). In the zeta potential measurements, 3.0 M HCl and 3.0 M NaOH were added in different flasks containing 30.0 mL distilled water to adjust the pH in the range of 0.70–6.00.

2.0 mg of adsorbent was added to 2.0 mL solution of each pH. The zeta potential of the solutions (0.8 mL) was measured. Zeta potentials of GO, N-GO, MgFe$_2$O$_4$NPs and nano-composite were measured at different pH (Fig. S12).

Both GO and N-GO formed stable dispersions in pH range from 2.20 to 6.14 with the highest value of zeta potential (-39.5 mV) for GO at pH 4.15 and -38.1 mV for N-GO at pH 4.11. In the acidic environment, with the decrease in pH, the COO$^-$ groups were gradually protonated which resulted in lower zeta potential value. The observed increase in zeta potential above pH 4.0 is due to compression of the double layer at high ionic strength. The zeta potential of GO was more negative at low pH due to the presence of COO$^-$ groups as compared to N-GO. The zeta potential of N-GO was lower due to a decrease in oxygen functional groups and an increase in the number of nitrogen functionalities on the surface. The zeta potential curves for MgFe$_2$O$_4$
NPs and nanocomposite displayed the isoelectric point at pH 3.0 and 4.3, respectively indicating the positive surface charge of NPs and nanocomposite at pH lower than these values and vice-versa. This may be due to large number of oxygen and nitrogen containing functional groups present on the surface of NPs and nanocomposite. At pH 6.0, surface of NPs and nanocomposite are negatively charged offering large number of active sites for adsorption of Pb(II) ions. Also, nanocomposite was more negatively charged thus indicating the presence of large number of surface functionalities resulting in higher Pb(II) removal efficiency as compared to MgFe₂O₄ NPs.

**Adsorption mechanism**

The adsorption mechanism involved both the surface complexation as well as electrostatic interaction of metal ions with different functional groups. There are large number of oxygen (O) and nitrogen (N) containing functional groups present on the surface of N-GO. Also, nanocomposite contained

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**Table 2** Langmuir isotherm parameters and statistical analysis for adsorption of Pb(II) and As(III) ions

| Isotherm model | Adsorbent | \( q_m \) (mg/g) | \( b_1 \times 10^{-3} \) (l/mg) | \( \chi^2 \) | \( G^2 \) |
|---------------|-----------|------------------|-------------------------------|-----------|---------|
| Langmuir      | GO        | 500.0 ± 4.5      | 11.7 ± 2.5                    | 0.96      | 0.018   |
|               | N-GO      | 690.0 ± 5.3      | 14.0 ± 4.0                    | 0.97      | 0.001   |
|               | MgFe₂O₄   | 740.7 ± 0.2      | 0.27 ± 0.4                    | 0.98      | 0.001   |
|               | MgFe₂O₄-NGO| 930.0 ± 0.5     | 0.32 ± 2.2                    | 0.97      | 0.004   |
| As(II)        | Go        | 22.7 ± 2.2       | 18.1 ± 1.7                    | 0.98      | 0.001   |
|               | N-GO      | 33.0 ± 3.5       | 23.9 ± 2.8                    | 0.97      | 0.002   |
|               | MgFe₂O₄   | 55.3 ± 0.5       | 0.09 ± 2.0                    | 0.99      | 0.003   |
|               | MgFe₂O₄-NGO| 64.1 ± 2.1      | 0.181 ± 1.3                   | 0.99      | 0.002   |

\( q_m \)-maximum adsorption capacity; \( b_1 \)-Langmuir constant; \( \chi^2 \)-regression coefficient; \( G^2 \)-Chi-square error; \( \chi^2 \)-log likelihood error

**Table 3** Comparison of the previously reported adsorbents for adsorption of Pb(II) and As(III) ions with the present investigation

| Adsorbent                        | pH  | Adsorbent dose (g/L) | Initial concentration (mg/L) | Adsorption capacity (qm) (mg/g) | Reference                                                                 |
|----------------------------------|-----|----------------------|-------------------------------|--------------------------------|---------------------------------------------------------------------------|
| Pb(II)                           |     |                      |                               |                                | Katubi et al. 2021)                                                       |
| MnFe₂O₄/GO                       | 6.0 | 0.16                 | 50.0                          | 960.0                          | Nik-Abdul-Ghani et al. 2021a; Nik-Abdul-Ghani et al. 2021b)               |
| GO-Polymer                       | 5.0 | 1.0                  | 100.0                         | 370.4                          | Tiwari and Kaur 2020)                                                     |
| SiO₂@MgFe₂O₄                     | 6.0 | 1.0                  | 10.0                          | 44.05                          | Parsa and Rezai 2021)                                                     |
| Melamine-modified nano GO        | 6.0 | 0.1                  | 200.0                         | 1896                           | Chowdhury et al. 2021)                                                    |
| Aluminium based metal organic framework-GO | 11.0 | 2.0                  | 90.0                          | 232.0                          | This study                                                                |
| N-GO                             | 6.0 | 1.0                  | 10.0                          | 689.6                          | This study                                                                |
| MgFe₂O₄-NGO                      | 6.0 | 1.0                  | 10.0                          | 930.0                          | This study                                                                |
| As(III)                          |     |                      |                               |                                |                                                                            |
| rGO-NiFe₂O₄                      | 6.5 | 0.3                  | 10.0                          | 65.78                          | Lingamdinne et al. 2016)                                                  |
| Polyether Sulfone-Graphene Oxide- Polyvinyl Pyrrolidone | 8.0 | 4.0                  | 55.0                          | 14,284.7                      | Nik-Abdul-Ghani et al. 2021a; Nik-Abdul-Ghani et al. 2021b                 |
| Graphene oxide-gadolinium oxide  | 6.0 | 0.1                  | 10.0                          | 216.7                          | Lingamdinne et al. 2021)                                                  |
| Chitosan-magnetic-GO             | 7.3 | 0.1                  | 10.0                          | 45.0                           | Sheralala et al. (2019)                                                    |
| N-GO                             | 7.0 | 2.0                  | 5.0                           | 33.0                           | This study                                                                |
| MgFe₂O₄-NGO                      | 7.0 | 2.0                  | 5.0                           | 64.1                           | This study                                                                |
unsaturated charges on the surface due to MgFe$_2$O$_4$ NPs in addition to N and O containing functionalities which are favorable for metal ion adsorption. The porous surface of nanocomposite as confirmed by BET studies supported the availability of greater surface-active sites for adsorption. Moreover, nanocomposite also contained M-OH and Fe-OH hydroxyl groups which are characteristic of MgFe$_2$O$_4$ NPs that act as main binding sites for various ionic species contributing to the variance in sorption capacity (Reddy and Yun 2016).

XPS analysis was also performed to support the adsorption results. The XPS spectrum of nanocomposite displayed the presence of N, C, O, Mg, and Fe (Fig. 8a). The C 1s XPS displayed the peak at 283.2 eV, which was assigned to the C–C bond with sp$^2$ hybridization. This confirmed the presence of graphite-like carbon. The peaks at binding energies of 398.2 eV, 399.2 eV, and 401.2 eV corresponded to the pyridinic N, graphitic N, and oxidic N, respectively (Yanilmaz et al. 2017)). The binding energies of C 1s and O 1s in XPS spectrum of Pb(II) and As(III) adsorbed MgFe$_2$O$_4$-NGO exhibited a slight positive shift as compared to the same nanocomposite before adsorption (Fig. 8b and c). This indicated the change in the local binding environment as the electrostatic forces may get disturbed due to the metal binding. After adsorption, the peaks of Pb 4f and As 3d appeared at the binding energies of 139.8 eV and 44.9 eV, respectively, demonstrating the presence of Pb(II) or As(III) ions on the surface of MgFe$_2$O$_4$-NGO. Lower binding energy for As(III)
ions further confirmed the lower affinity of nanocomposite for As(III) ions as compared to Pb(II) ions.

The FT-IR spectra of Pb(II) and As(III) adsorbed MgFe$_2$O$_4$-NGO are given in Fig. S13. For Pb(II) adsorbed MgFe$_2$O$_4$-NGO, the peak of O–H and N–H groups at 3153 cm$^{-1}$ shifted to 3124 cm$^{-1}$ which may be attributed to the interaction of Pb(II) with oxygen and nitrogen functional groups. Both oxygen and nitrogen contain lone pair of electrons which can bind to Pb(II) by sharing the electrons and form stable complex. Also, the peak at 1558 cm$^{-1}$ for N–H bond shifted to 1575 cm$^{-1}$ indicating the formation of Pb(II)-N complex (Zhou et al. 2014). The peak at 1387 cm$^{-1}$ and 1199 cm$^{-1}$ shifted to 1399 cm$^{-1}$ and 1205 cm$^{-1}$, respectively which may be due to ion exchange between Pb(II) and N–H group. In case of As(III) adsorbed MgFe$_2$O$_4$-NGO, a new peak appeared at 701 cm$^{-1}$ which corresponded to the stretching vibration of uncomplexed As(III)-O bond. This confirmed loosely bound As(III) species. Similar results were reported by Pena et al. for arsenic adsorption of TiO$_2$ with the peak at 762 cm$^{-1}$ (Pena et al. 2006). Also, the M–O stretching bands of ferrite NPs did not alter. indicating the electrostatic interactions and physisorption to be the suitable mechanism for adsorption.

**Effect of co-existing ions on adsorption**

Competition of ubiquitous anions present in natural water with Pb(II) and As(III) ions for adsorptive sites was studied for three selected anions, i.e., Cl$^-$, CO$_3^{2-}$, and PO$_4^{3-}$ ions. As shown in Fig. 9a and b, Cl$^-$ had no significant effect on adsorption of both Pb(II) and As(III) ions. Comparatively, CO$_3^{2-}$ and PO$_4^{3-}$ ions affected the removal of Pb(II) and As(III) ions. The presence of CO$_3^{2-}$ ion decreased the adsorption of As(III) ions to a greater extent compared to that of Pb(II) ions. This decrease may be due to the formation of arseno-carbonate complexes, including As(CO$_3$)(OH)$_2^{2-}$ and As(CO$_3$)$_2^{2-}$, at high concentrations of CO$_3^{2-}$, thus preventing As(III) ions from coordinating to the adsorbent surface (Su and Puls 2001)). In the case of Pb(II) ions, the removal was slightly decreased on increasing the concentration of CO$_3^{2-}$ ions due to the formation of a lead-carbonate complex which prevents the adsorptive removal of Pb(II) ions (Kong et al. 2017). It is noteworthy that the adsorption of Pb(II) and As(III) ions was greatly affected by phosphate ions. As the concentration of phosphate ions increased, the adsorption capacity of MgFe$_2$O$_4$@N-GO nanocomposite was decreased from 93.5 ± 0.1% to 81.1 ± 9.7% for As(III) ions and from 99.5 ± 0.8% to 92.0 ± 5.2% for Pb(II) ions. The greater decrease in adsorption of As(III) ions might result from the strong competition between phosphate ions (e.g., PO$_4^{3-}$) and As(III) ions towards the adsorption sites as compared to Pb(II) ions (Zheng et al. 2008)).

The influence of common existing cations on Pb(II) adsorption is presented in Fig. 9c. It shows the removal of different heavy metal ions from 100 mL of Cd–Pb-Ni-Zn multi-ion solution. The percentage removal of Pb(II) ions was significantly affected by the presence of coexisting ions. Metal ion removal trend was Pb(II) > Cd(II) > Zn(II) > Ni(II). Thus, Pb(II) ions were preferentially adsorbed on the surface of the adsorbent as compared to other ions in multi ion system. This trend can be explained on the basis of the ionic radii of metal ions with the trend: Pb(II) > Cd(I I) > Zn(II) > Ni(II) where increase in ionic radii resulted in decreased hydration capacity which further led to smaller hydrated ionic radii and hence more Pb(II) adsorption (Fan et al. 2008)).
Electroplating effluent tests

Industrial water is much more complex than deionized water due to presence of different heavy metal ions in it. The Pb(II) removal from industrial effluent was studied by making the solution basic using 1.0 N NaOH in order to precipitate out metal ions. This was followed by dilution of wastewater (five times) before the treatment with adsorbent to bring concentration of metal ion in the studied range for the single component system. The industrial waste water contained initial concentration of metal ions as: 5.13 mg/L Ni(II), 0.05 mg/L Pb(II), 0.28 mg/L As(III), 0.24 mg/L Zn(II), 0.01 mg/L Cd(II), 0.32 mg/L Mg(II), 0.26 mg/L Ca(II), and 1.40 mg/L B (III). Fifty milliliters of the above solutions and 0.10 g of the adsorbent were tested for their efficiency at 25 ºC. After treatment with MgFe$_2$O$_4$-NGO nanocomposite, the Pb(II) ion removal was observed to be 78.20 ± 0.2% (Fig. S14a). However, the percentage removal from industrial effluent was less than spiked Pb(II) water as the industrial waste water contained As(III) and other interfering ions which was less than spiked Pb(II) water as the industrial waste water contained As(III) and other interfering ions which were preferentially adsorbed followed by Cd(II), Zn(II), and Ni(II) ions. The nanocomposite also displayed effectiveness in treating the industrial wastewater. Thus, nanofabrication of N-GO with MgFe$_2$O$_4$ NPs was observed to be an effective strategy to enhance the adsorption potential of ferrite NPs for water decontamination. It can be explored as a potential adsorbent for other organic and inorganic contaminants.

Recyclability of nanocomposite

Desorption and reuse of the adsorbents may reduce the cost of operation and open the possibility of recovering the metals extracted from the liquid phase ( Harinath et al. 2017). Six adsorption–desorption cycles were performed to test the reusability of nanocomposite as an adsorbent for Pb(II) and As(III) removal (Fig. S14b). Initially, the removal efficiency of nanocomposite for Pb(II) ions was ≈ 99.5 ± 0.3%. After the sixth adsorption–desorption cycle, the adsorption efficiency of nanocomposite for Pb(II) ions was measured to be ≈ 89.1 ± 0.2%. In the case of As(III) ions, nanocomposite showed a removal efficiency of 93.5 ± 0.8%, which reduced to 83.6 ± 1.0% after the sixth adsorption–desorption cycle. These results suggested the promising reusability and stability during the repeated use of nanocomposite for the adsorption of Pb(II) and As(III) ions.

Conclusions

Nanofabrication of N-GO with MgFe$_2$O$_4$ NPs resulted in higher adsorption capacity of nanocomposite for Pb(II) and As(III) ions as compared to pristine NPs, GO and N-GO. Different techniques fully characterized the synthesized adsorbents, which assisted in understanding the adsorption mechanism. Higher porosity of the nanocomposite than the pristine NPs and N-GO resulted in enhanced removal efficiency of 99.7 ± 0.2% and 93.5 ± 0.1% for Pb(II) and As(III) ions, respectively. The Langmuir model could fit adsorption results reasonably, which caused significantly high monolayer adsorption capacity (qm) of the nanocomposite as 930.0 mg/g and 64.1 mg/g for Pb(II) and As(III), respectively. The adsorption results were well supported by XPS studies. In the presence of other heavy metal ions, Pb(II) ions were preferentially adsorbed following Cd(II), Zn(II), and Ni(II) ions. The nanocomposite also displayed effectiveness in treating the industrial wastewater. Thus, nanofabrication of N-GO with MgFe$_2$O$_4$ NPs was observed to be an effective strategy to enhance the adsorption potential of ferrite NPs for water decontamination. It can be explored as a potential adsorbent for other organic and inorganic contaminants.
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