Removal of Cu$^{2+}$, Zn$^{2+}$ and Pb$^{2+}$ ions using a graphene-containing nanocomposite: a kinetic study

A Babkin$^1$, I Burakova$^1$, A Burakov$^1$, D Kurnosov$^1$, E Galunin$^1$, A Tkachev$^1$ and I Ali$^2$

$^1$Department of Technology and Methods of Nanoproducts Manufacturing, Tambov State Technical University, 106 Sovetskaya Str., Tambov 392000, Russian Federation
$^2$Department of Chemistry, Jamia Millia Islamia (Central University), Jamia Nagar, New Delhi 110025 India

$^1$E-mail: iris_tamb68@mail.ru

Abstract. In the present paper, the urgent issue of removing heavy metal ions from aqueous solutions is considered by the example of extraction of Cu$^{2+}$, Zn$^{2+}$ and Pb$^{2+}$ ions using an innovative sorbent - polyhydroquinone/graphene nanocomposite. Kinetics of the metal sorption on this material was studied. Moreover, the effect of solution pH on the sorption capacity of the nanocomposite was estimated. Besides, to elucidate mechanisms of kinetic removal and determine process parameters, experimental data were fitted to kinetic (pseudo-first- and pseudo-second-order, and Elovich) and diffusion (intraparticle and Boyd) models. The kinetic experiments showed that equilibrium is reached within 60 min. The effective diffusion coefficient values calculated according to the Boyd model were found to increase in the following sequence: Pb$^{2+}$ (1.26 $10^{-11}$ m$^2$ s$^{-1}$) < Zn$^{2+}$ (3.74 $10^{-11}$ m$^2$ s$^{-1}$) < Cu$^{2+}$ (4.24 $10^{-11}$ m$^2$ s$^{-1}$). Finally, it was established that the metal sorption process is controlled by the internal diffusion of the sorbate into the bulk (volume) of the sorbent.

1. Introduction

In the field of highly efficient purification of aquatic systems, many scientific papers prove that carbon nanostructured materials are more effective compared to currently used commercially available sorbents [1-4]. Moreover, they are non-toxic and thermally stable. The prospect of using such nanomaterials in sorption-desorption processes is justified, on the one hand, by their unique properties (such as specific surface area, presence of surface chemically active sites, high chemical affinity for components being extracted), and, on the other hand, by the similarity of the chemical nature (allotropy) and a set of mechanical characteristics to those of the most widely used industrial sorbents [5, 6]. This will allow them to be employed without doing any harm to the microelemental and mineral composition of soils and aquatic media. In the near future, the proposed carbon nanostructured sorbents will become an alternative to conventional sorption materials based on rocks, clays, hydrated aluminosilicates, activated carbons, low-cost agro-waste biosorbents, etc.

Among the existing physical and chemical processes of removal and subsequent isolation of harmful contaminants, sorption-desorption cycles play a significant role. Only their practical implementation makes it possible to extract, transfer and accumulate toxic materials for appropriate storage conditions. In this regard, elucidating fundamentals and empirical dependencies of selective
extraction of various harmful contaminants from target aquatic and terrestrial systems using nanostructured sorbents is an important task. Although a number of scientific studies (table 1) have shown the promising use of carbon graphene-containing structures (multi- (MWCNTs) and single-walled (SWCNTs) carbon nanotubes, graphene oxide (GO)) as sorbents for the extraction of such heavy metals as Fe, Cu, Ni, Zn, Cr, Pb, etc [6-8], to the best of our knowledge, mechanisms of metal removal processes occurring through sorption-desorption cycles have not completely and reliably studied yet so far.

**Table 1.** Comparative characteristics of the adsorption of Cu$^{2+}$, Pb$^{2+}$, Zn$^{2+}$ on various carbon nanomaterials.

| Adsorbent | Experimental conditions | pH | Temperature, K | Initial concentration, mg L$^{-1}$ | Adsorbent weight, g | Time, min | Q$^e$, mg·g$^{-1}$ |
|-----------|------------------------|----|----------------|-----------------------------------|---------------------|-----------|----------------|
| Pb$^{2+}$ |                        |    |                |                                   |                     |           |                |
| MWCNTs (oxidized) [9] | 5 | room temperature | 40 | 1 | 150 | 50 |
| GO [10] |                        | 5 | room temperature | - | 0.02 | 90 | 44.56 |
| rGONF (ferrite-reduced GO) Nanocomposite [10] | 5 | room temperature | - | 0.02 | 30 | 25.78 |
| Mn$_3$O$_4$-coated activated carbon [11] | 5 | 303 | 50 | 1 g·L$^{-1}$ | 70 | 49.82 |
| Cu$^{2+}$ |                        |    |                |                                   |                     |           |                |
| Mn$_3$O$_4$-coated activated carbon [11] | 5 | 303 | 50 | 1 g·L$^{-1}$ | 70 | 25.66 |
| GO/MWCNTs/Fe$_3$O$_4$ nanocomposite [12] | 7 | room temperature | 25 | 25 | - | 20.83 |
| GO [13] |                        | 6 | room temperature | 100 | 0.03 | 20 | 33 |
| Zn$^{2+}$ |                        |    |                |                                   |                     |           |                |
| SWCNTs [14] | 7 | room temperature | 60 | 0.05 | 360 | 39.52 |
| Oxidized MWCNTs [15] | 5 | room temperature | 3 mmol L$^{-1}$ | 0.005 | 120 | 0.2 mmol g$^{-1}$ |
| GO [16] |                        | 6 | room temperature | 100 | 0.03 | 20 | 26 |

Considering the aforementioned, the aim of the present work was to study sorption regularities using the kinetic approach, which allows estimating the effect of system parameters, varying over time, on the degree of extraction of target products from model systems and their retention. A novel polyhydroquinone/graphene (PHQ/G) nanocomposite was employed as a sorbent. The results of the work will contribute to the expansion of the application of the developed technologies in wastewater treatment programs in populated areas, where aquatic systems are contaminated with heavy metals and toxic organic compounds due to the abundance of industrial effluents.
2. Experimental part

2.1. Material and reagents

The PHQ/G nanocomposite was produced by NanoTechCenter Ltd (Tambov, Russia). The technology of its manufacturing is described elsewhere [17]. Heavy metal salts – Pb(NO\textsubscript{3})\textsubscript{2}, Cu(NO\textsubscript{3})\textsubscript{2} and Zn(NO\textsubscript{3})\textsubscript{2} (all – analytical grade) were acquired from Laverna Lab Ltd (Moscow, Russia). Heavy metal stock solutions were obtained by diluting certain amounts of the corresponding salts in deionized water. Before using in tests, the solutions diluted to the required concentrations.

2.2. pH effect on the sorption capacity

pH is an important parameter, since it determines the number of acid-base groups, which, in turn, affects the degree of interaction between the contaminant and the surface groups of the sorbent. To study its effect on the removal sorption (removal) capacity of the PHQ/G nanocomposite for Pb\textsuperscript{2+}, Cu\textsuperscript{2+} and Zn\textsuperscript{2+} ions, the following experiment was performed at room temperature.

0.03 g of the PHQ/G nanocomposite were placed into 50-mL conical tubes. Then, 30 mL of the metal solution at a preset concentration and pH value was poured into the tubes. Prior to that, the solution pH was fixed at the predetermined value using a buffer system obtained by adding corresponding acids (hydrochloric, acetic, and aminoacetic) and alkali (sodium hydroxide) in certain proportions. The pH of each solution was measured using an HI 2210 pH meter (Hanna Instruments, Woonsocket, RI, USA). The tubes were shaken for 60 min on a Multi Bio RS-24 programmable rotator (Biosan, Riga, Latvia), and then filtered to separate the solid phase. The equilibrium metal concentration in the selected aliquots was measured using an MGA-915 MD atomic absorption spectrometer (Atomprбор, St. Petersburg, Russia).

The equilibrium sorption capacity (\(Q_e\), mg g\(^{-1}\)) of the material was determined according to equation (1):

\[
Q_e = \frac{V(C_0 - C)}{m},
\]

where \(C_0\) – initial metal concentration in solution (before sorption), mg L\(^{-1}\); \(C\) – equilibrium metal concentration in solution (after sorption), mg L\(^{-1}\); \(V\) – solution volume, mL; \(m\) – sorbent weight, g [18].

2.3. Kinetic study

To identify the features of the Cu\textsuperscript{2+}, Pb\textsuperscript{2+} and Zn\textsuperscript{2+} sorption on the PHQ/G nanocomposite, batch kinetic tests were carried out. The purpose was to establish the time to reach equilibrium (at which the sorption and the desorption rates are equal) in the sorbate-sorbent system under specific process conditions, and to describe the mechanism of metal ion sorption.

0.01 g of the PHQ/G nanomaterial and 30 mL of 100-mg L\(^{-1}\) were put into 50-mL tubes, and then shaken on the rotator (Biosan) for 60 min. The average number of experimental samples to be analyzed was 13 (i.e., it corresponded to sampling time intervals). Due to the high sorption activity found for the material during the initial period of the process, it was necessary to pick the maximum possible number of samples – after 10, 20, 30 and 60 s. Further aliquot sampling intervals were 1, 2, 5, 7, 10, and 15 min. After this time, the sorption activity of the nanocomposite usually decreases significantly, and thus, the sampling was carried out after 30, 45, and 60 min.

To estimate parameters and describe possible kinetic mechanisms of the metal sorption on the PHQ/G nanocomposite, the following models were implemented (Table 2): kinetic – pseudo-first- and pseudo-second-order, and Elovich, and diffusion – intraparticle and Boyd [18].
Table 2. Kinetic and diffusion models implemented [18].

| Model                        | Equation                    |
|------------------------------|-----------------------------|
| Pseudo-first-order           | \( \log(Q_e - Q_t) = \log(Q_e) - \frac{k_1 t}{2.303} \) (2) |
| Pseudo-second-order          | \( \frac{t}{Q_t} = \frac{1}{k_2 Q_e^2} + \frac{1}{Q_e} \cdot t \) (3) |
| Elovich                      | \( Q_t = \frac{1}{\beta} \ln(a\beta) + \frac{1}{\beta} \ln(t) \) (4) |
| Intraparticle diffusion      | \( Q_t = k_{id} \cdot t^{0.5} + C \) (5) |
| Boyd                         | \( F = 1 - \frac{6}{\pi^2} \sum_{n=1}^{\infty} \frac{1}{n^2} \exp(-Bn^2), \) \( B = \frac{Dn^2}{r_0^2} \) (7) |

Parameters of equations (2)-(7): \( t \) – sorption time, min; \( Q_e \) – equilibrium sorption capacity, mg g\(^{-1}\); \( Q_t \) – sorption capacity at time \( t \), mg g\(^{-1}\); \( k_1 \) – pseudo-first-order reaction rate constant (min\(^{-1}\)); \( k_2 \) – pseudo-second-order reaction rate constant, g mg\(^{-1}\) min\(^{-1}\); \( a \) – initial flow rate, min\(^{-1}\) mg g\(^{-1}\); \( \beta \) – desorption constant (degree of surface coverage and activation energy of chemisorptions), g mg\(^{-1}\); \( k_{id} \) – intraparticle diffusion coefficient, min\(^{-1/2}\); \( c \) – boundary layer thickness constant, mg g\(^{-1}\); \( F \) – degree of process completion; \( n \) – number of sum items; \( D \) – effective diffusion coefficient (m\(^2\) s\(^{-1}\)); \( r_0 \) – average radius of the sorbent grain, m.

3. Results and discussion

The results of the pH effect experiments are presented in figure 1.

![Figure 1. The aqueous solution pH effect on the Pb\(^{2+}\), Cu\(^{2+}\) and Zn\(^{2+}\) sorption capacity of the PHQ/G nanocomposite (conditions: sorbent weight – 0.03 g, temperature – 298 K, and time – 60 min).](image)

From this figure, it can be seen that the maximum sorption characteristics of the nanocomposite being studied are achieved in the solutions at pH=6 (the dependences have extrema at this point). An increase in the degree of extraction of Pb\(^{2+}\), Cu\(^{2+}\) and Zn\(^{2+}\) ions with an increase in pH indicates that these ions are in optimal interaction condition. Relatively low sorption capacity values in the acidic medium (at pH = 2 and 4) are probably due to the protonation of the sorbent, which leads to the displacement of Pb\(^{2+}\), Cu\(^{2+}\) and Zn\(^{2+}\) ions (they may be available in solution as aqua complexes [16]). In the pH range of 3-6, the metal sorption capacity increases and reaches the maximum value at pH=6. In
this area, the functional groups of the nanocomposite are ionized, and the existing aqua complexes may be dissociated. Further increasing the solution pH leads to a decrease in the sorption capacity of the material. Based on the results of this test, the other experiments were carried out in buffer solutions at the optimum pH value (i.e., pH = 6).

Figure 2 shows the graphical dependences of the sorption capacity of the PHQ/G nanocomposite and the reference materials in their original form – carbon nanotubes (CNTs), graphene oxide (GO), and activated carbon (AC) on the phase contact time.

![Graphical representation of sorption capacity](image)

**Figure 2.** Kinetic curves constructed for the metal sorption on the PHQ/G nanocomposite in comparison with the reference materials (CNTs, GO and AC) in the time ranges of 0-60 min (a) and 0-5 min (b) (conditions: sorbent weight – 0.01 g, initial metal concentration in solution – 100 mg L^{-1}, temperature – 298 K, and pH=6).

It can be seen that the sorption capacity of the PHQ/G material is 2-25 times higher than that of the reference materials. It should be noted that the nature of the metal sorption on the CNTs, GO and AC differs from process taking place on the nanocomposite, which is characterized by a sharp increase in the sorption capacity during the initial period of time – about 70 % of the sorbate is sorbed within the first 5 min of the process (Figure 2b). The sorbate/sorbent system reaches equilibrium 60 min after the sorption starts.

Besides, it was experimentally found that the maximum sorption capacity ($Q_{e,exp}$) of the PHQ/G nanocomposite is as follows: ~174, ~97 and ~64 mg g^{-1} regarding Pb^{2+}, Cu^{2+} and Zn^{2+}, respectively.

Figures 3-5 and Tables 3 and 4 demonstrate the fitting of the experimental data, obtained for the sorption kinetics of all the metal ions, to the kinetic (pseudo-first-order, pseudo-second-order, and Elovich) and diffusion (intraparticle) models (for the Boyd diffusion model, the graphs are not shown; the parameters of this model were obtained using the computer algebra system Maxima).
Figure 3. Cu$^{2+}$ sorption data fitted to the pseudo-first-order (a), pseudo-second-order (b), Elovich (c) and intraparticle diffusion (d) models.
Figure 4. Pb$^{2+}$ sorption data fitted to the pseudo-first-order (a), pseudo-second-order (b), Elovich (c) and intraparticle diffusion (d) models.
Figure 5. Zn$^{2+}$ sorption data fitted to the pseudo-first-order (a), pseudo-second-order (b), Elovich (c) and intraparticle diffusion (d) models.

The high correlation of the experimental data to the pseudo-first-order model, judged by the coefficients of determination ($R^2$) being equal to 0.941 (Cu$^{2+}$), 0.945 (Zn$^{2+}$) and 0.952 (Pb$^{2+}$), mainly in the initial period of time (up to 7 min – for Cu$^{2+}$, up to 30 min – for Pb$^{2+}$, and up to 2 min – for Zn$^{2+}$), evidences the occurrence of chemical interactions between the active functional groups of the sorbent and the heavy metal ions.

The pseudo-second-order model satisfactorily describes the removal process for all the ions ($R^2$ = 0.999 (Cu$^{2+}$ and Zn$^{2+}$) and 0.998 (Pb$^{2+}$)). The correspondence of the experimental data to this model indicates that the sorbate–sorbent chemical reaction (i.e., pseudo-second-order reaction) takes place. It should be noted that the calculated $Q_e$ values coincide with experimental $Q_{e,exp}$ values for all the heavy metal ions. At the same time, the pseudo-second-order reaction rate constant is on average 10-50 times lower than during the interaction between the sorption sites of the material and the metal ions.

The correspondence of the data to the Elovich model ($R^2$ = 0.989 (Cu$^{2+}$), 0.983 (Zn$^{2+}$) and 0.953 (Pb$^{2+}$)) suggests that the sorbent has high-energy heterogeneous surfaces [18]. This leads to very rapid sorption of the solute by the sorbent until the system stabilizes.
G R S 2 0 1 9

I O P C o n f . S e r i e s : M a t e r i a l s S c i e n c e a n d E n g i n e e r i n g 6 9 3 ( 2 0 1 9 ) 0 1 2 0 2 9
doi : 1 0 . 1 0 8 8 / 1 7 5 7 - 8 9 9 X / 6 9 3 / 1 / 0 1 2 0 2 9

Table 3. Parameters of the kinetic models obtained for the metals sorption on the PHQ/G nanocomposite.

| Sorbate | Pseudo-first-order | Pseudo-second-order | Elovich |
|---------|---------------------|---------------------|---------|
|         | Qc, mg g⁻¹ | k₁, min⁻¹ | R² | Qc, mg g⁻¹ | k₂, g mg⁻¹ | R² | α, min⁻¹ | β, g mg⁻¹ | R² |
| Cu²⁺    | 68.6     | 0.336       | 0.941 | 99.4     | 0.0210 | 0.999 | 1809.1  | 0.083   | 0.989 |
| Zn²⁺    | 74.13    | 0.808       | 0.945 | 71.6     | 0.0183 | 0.999 | 1005.4  | 0.109   | 0.983 |
| Pb²⁺    | 119.6    | 0.165       | 0.952 | 178.6    | 0.0034 | 0.998 | 337.3   | 0.033   | 0.953 |

Table 4. Parameters of the diffusion models obtained for the metals sorption on the PHQ/G nanocomposite.

| Sorbate | Intraparticle diffusion | Boyd |
|---------|-------------------------|------|
|         | Line 1 | Line 2 | Line 3 | B | D | R² |
| kₐᵣ, min⁻¹/² | C₂, mg g⁻¹ | R² | kₜᵣ, min⁻¹/² | C₃, mg g⁻¹ | R² | kₒᵣ, min⁻¹/² | C₄, mg g⁻¹ | R² | 10⁻⁻¹ m² s⁻¹ |
| Cu²⁺    | 51.8   | 7.348  | 0.948 | 12.9 | 59.05 | 0.938 | 0.27 | 96.41 | 0.949 | 0.00865 | 4.24 | 0.980 |
| Zn²⁺    | 60.7   | 3.267  | 0.887 | 9.86 | 34.95 | 0.974 | 1.17 | 62.5 | 0.686 | 0.00763 | 3.74 | 0.880 |
| Pb²⁺    | 68.2   | 4.448  | 0.983 | 29.5 | 50.36 | 0.981 | 2.35 | 156.5 | 0.886 | 0.00264 | 1.29 | 0.980 |

The metal diffusion penetration effect on the overall process rate was elucidated by implementing the intraparticle diffusion model (R²=0.938-0.949 (Cu²⁺), 0.686-0.974 (Zn²⁺) and 0.886-0.983 (Pb²⁺)). It is clear that the graphs are non-linear over the entire time interval, which means that the sorption process is limited by more than one stage. The intraparticle diffusion sorption model has three intersecting lines for each ion. The first and the second lines are associated with surface sorption and intraparticle diffusion, whereas the third line represents the final stage of equilibrium, at which the intraparticle diffusion begins to slow down due to the extremely low concentration of the sorbate remaining in the solution. However, for all the ions, the graphs leave the origin, therefore, the internal diffusion limits the total time of the sorption process.

To confirm this assumption, the Boyd model (R²=0.980 (Cu²⁺ and Pb²⁺) and 0.880 (Zn²⁺)) was implemented in the initial period of the process. Due to the mathematical processing of the experimental data, the effective diffusion coefficients characterizing the rate of sorbate ion penetration inside the sorbent grains were calculated. They were found to increase in the following sequence: Pb²⁺ (1.29 10⁻¹¹ m² s⁻¹) < Zn²⁺ (3.74 10⁻¹³ m² s⁻¹) < Cu²⁺ (4.24 10⁻¹³ m² s⁻¹).

Thus, the values obtained for the adsorption capacity of the PHQ/G nanocomposite regarding the Cu²⁺, Pb²⁺ and Zn²⁺ ions are significantly higher in comparison with the ones obtained for the other materials listed in table 1 (CNTs, GO, modified activated carbon). This fact makes it possible to judge on the feasibility of using the PHQ/G nanocomposite in the processes of liquid-phase sorption of heavy metal ions.

4. Conclusion

As a result of the experimental studies on the sorption of Cu²⁺, Zn²⁺ and Pb²⁺ ions onto the PHQ/G nanocomposite material, it was found that the equilibrium is reached within 60 min. The following maximum sorption capacity of the sorbent was achieved: ~174, ~97 and ~64 mg g⁻¹ regarding Pb²⁺, Cu²⁺ and Zn²⁺, respectively.

Using the pseudo-first and second-order, Elovich and intra-diffusion models, it was found that the sorption is limited by the internal diffusion of the metal ions into the porous space of the PHQ/G nanocomposite, taking into account the contribution of the chemical interaction between the metal ions and the functional groups of the sorbent during the initial period of the process (according to the pseudo-second order model).
Acknowledgment
The research was supported by the Ministry of Science and Higher Education of the Russian Federation under Project No. 16.1384.2017/PCh.

References
[1] Hua M, Zhang S, Pan B, Zhang W, Lv L and Zhang Q 2012 Heavy metal removal from water/wastewater by nanosized metal oxides, a review J. Hazard. Mater. 211-212 317–331
[2] Ashour R W, Abdelhamid H N, Abdel-Magied A F, Abdel-Khalek A A, Ali M M, Uheida A, Muhammed M, Zhou X and Dutta J 2017 Rare earth ions adsorption onto graphene oxide nanosheets Solvent Extr. Ion Exch. 35 91–103
[3] Harvey P J, Handley H K and Taylor M P 2015 Identification of the sources of metal (lead) contamination in drinking waters in north-eastern Tasmania using lead isotopic compositions Environ. Sci. Pollut. Res. 22 12276–12288
[4] Fu F and Wang Q 2011d Removal of heavy metal ions from wastewaters, a review J. Environ. Manag. 92 407–418
[5] Gu D and Fein J B 2015 Adsorption of metals onto graphene oxide, surface complexation modeling and linear free energy relationships Colloids Surf. A: Physicochem Eng. Aspects 481 319–327
[6] Dreyer D R, Park S, Bielawski C W and Ruoff R S 2010b The chemistry of graphene oxide Chem. Soc. Rev. 39 228–240
[7] Kumar R, Chawla J and Kaur I 2015 Removal of cadmium ion from wastewater by carbon-based nanosorbents, a review J. Water Health 13 19–33
[8] Duru I, Ege D and Kamali A R 2016 Graphene oxides for removal of heavy and precious metals from wastewater. J. Mater. Sci. 51 6097–6116
[9] Wang H J, Zhou A L, Peng F, Yu H and Chen L F 2007 Adsorption characteristic of acidified carbon nanotubes for heavy metal Pb(II) in aqueous solution Mater. Sci. Engin. A. 466 201-206
[10] Lingamdinne L P, Kim I S, Ha J H, Chang Y Y, Koduru J R and Yang J K 2017 Enhanced Adsorption Removal of Pb(II) and Cr(III) by Using Nickel Ferrite-Reduced Graphene Oxide Nanocomposite Metals 7 1225-237
[11] Lee M E, Park J H, Chung J W, Lee C Y and Kang S 2015 Removal of Pb and Cu ions from aqueous solution by Mn3O4-coated activated carbon J. Indus. Engin. Chem. 21 470-475
[12] Long Z, Zhan Y, Li F, Wan X, He Y, Hou C and Hu H 2017 Hydrothermal synthesis of graphene oxide/multiwalled carbon nanotube/Fe3O4 ternary nanocomposite for removal of Cu (II) and methylene blue J. Nanoparticle Res. 19(9) 318-329
[13] Babkin A V, Burakov I V, Burakova A E, Neskoromnaya E A, Kucherova A E and Kurnosov D A 2018 Kinetics of the Cu(II) sorption from aqueous solutions by carbon nanomaterials Nanosystems: physics, chemistry, mathematics 9(1) 117-119
[14] Lu C and Chiu H 2006 Adsorption of zinc (II) from water with purified carbon nanotubes Chem. Engin. Sci. 61 1138-1145
[15] Cho H H, Wepasnick K, Smith B A, Bangash F K, Fairbrother D H and Ball W P 2010 Sorption of aqueous Zn(II) and Cd(II) by multiwall carbon nanotubes: the relative roles of oxygen-containing functional groups and graphenic carbon Langmuir 26 967–981
[16] Galunin E V, Burakova I V, Neskoromnaya E A, Babkin A N, Melezzhik A V, Burakov A E and Tkachev A G 2018 Adsorption of heavy metals from aqueous media on graphene-based nanomaterials AIP Conference Proceedings 2041 020007
[17] Babkin A V, Melezzhik A V, Kurnosov D A, Mkrtchyan E S, Burakova I V, Burakov A E and Galunin E V 2018 Kinetics of the Adsorption of Synthetic Dyes on a Polyhydroquinone/Graphene Carbon Nanocomposite J. Phys.: Conf. Ser. 1124 081030
[18] Gautam R K and Chattopadhyaya M Ch 2016 Nanomaterials for Wastewater Remediation (Oxford: Elsevier) p 353