

Energy-Efficient and Sustainable Electrochemical Technologies for Heavy Metal Removal and Recovery from Wastewater: A Critical Review (2000–2026)

Surinder Deswal*

Department of Civil Engineering, National Institute of Technology Kurukshetra, Kurukshetra, India.
*Corresponding Author Email: sdeswal@nitkkr.ac.in

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Abstract

Electrochemical methods are attracting growing interest for treating wastewater that contains heavy metals. They need few added chemicals, work at room temperature and pressure, and often allow the metal to be recovered in a usable form. How sustainable they are depends mainly on their electricity use, the rate at which electrodes are consumed and the handling of the sludge or concentrate produced, and these aspects are reported very differently from one study to the next. This review examines electrochemical methods for removing and recovering heavy metals with energy use and sustainability as the main points of comparison, using adsorption and pressure-driven membranes as reference technologies. English-language literature published between January 2000 and June 2026 was identified through a structured, targeted search of the literature. Five groups of processes are discussed: electrocoagulation, electrodeposition and electroreduction, capacitive deionization and electrosorption, electrodialysis and electrodeionization, and bioelectrochemical systems. Solar-powered systems, alternating or pulsed current operation and hybrid process trains are treated separately. For each group, the factors that control energy demand are examined, including cell voltage, current efficiency, mass transfer, electrode passivation and solution conductivity. A common way of reporting energy is proposed, based on energy per cubic metre treated, energy per kilogram of metal removed and comparison with the theoretical minimum. Alternating or pulsed current, polarity reversal, three-dimensional electrodes, membrane and flow-electrode capacitive designs and direct solar coupling have all been shown to reduce energy demand or operating cost, and bioelectrochemical cathodes can reduce some metals while producing electricity. Most studies, however, still use synthetic single-metal solutions, short batch experiments and incomplete energy accounting, and few include life cycle or cost analysis. Standard reporting, longer pilot trials on real effluents, reuse of residues and better process control are the main research needs.

Keywords: *Electrocoagulation; Capacitive deionization; Specific energy consumption; Metal recovery; Bioelectrochemical systems; Circular economy;*

INTRODUCTION

Arsenic, cadmium, chromium, copper, lead, mercury, nickel and zinc are among the most common and persistent inorganic pollutants found in industrial wastewater. These elements do not break down in the environment; they accumulate in living organisms and along food chains, and several of them are toxic or carcinogenic even at low concentrations (Järup, 2003; Tchounwou *et al.*, 2012; Jaishankar *et al.*, 2014). Electroplating and metal finishing, mining, tanning, battery manufacture, electronics, and smelting are the main industrial sources. In addition, arsenic of natural origin contaminates groundwater that is used for drinking by large populations in several parts of the world (Smedley and Kinniburgh, 2002; Schwarzenbach *et al.*, 2010). Stricter discharge limits and the rising value of metals recovered from waste have changed the aim of treatment. Plants are now expected not only to meet discharge standards but, where possible, to recover the metal as well (Vardhan *et al.*, 2019; Yang *et al.*, 2021).

The conventional treatment options are chemical precipitation, coagulation and flocculation, ion exchange, adsorption, and membrane filtration. Their advantages and drawbacks have been discussed in many reviews (Kurniawan *et al.*, 2006; Barakat, 2011; Fu and Wang, 2011; Carolin *et al.*, 2017; Azimi *et al.*, 2017). Precipitation is simple and cheap, but it produces large volumes of metal-bearing sludge

and is not very effective when metal concentrations are below about 1 mg/L. Adsorption is easy to operate, but the adsorbent has to be regenerated with acid or alkali, which creates a secondary waste stream (Babel and Kurniawan, 2003; Burakov *et al.*, 2018). Reverse osmosis and nanofiltration remove metals well, but they consume energy to generate pressure and suffer from membrane fouling and the problem of disposing of the reject stream (Abdullah *et al.*, 2019; Qasem *et al.*, 2021).

Electrochemical processes use electric current, in effect electrons, in place of chemical reagents. Depending on the process, the current is used to generate a coagulant inside the reactor, to reduce dissolved metal to metallic form, or to move ions through membranes or into the pores of charged electrodes (Chen, 2004; Comninellis and Chen, 2010; Sillanpää and Shestakova, 2017). Such processes are well suited to small, decentralised and automated plants, and they can be run directly from solar panels (Radjenovic and Sedlak, 2015; Chaplin, 2019). Their main weakness is electricity consumption. In electrocoagulation, for example, energy use rises sharply with current density, with the gap between electrodes and with the resistance of the solution, and oxide films on the electrodes add further losses (Mollah *et al.*, 2004; Moussa *et al.*, 2017; Khan *et al.*, 2023). Electricity is therefore the largest factor in both the running cost and the environmental impact of these processes.

Many earlier reviews have dealt with electrochemical removal of heavy metals. Some give a general account of electrocoagulation (Mollah *et al.*, 2001; Holt *et al.*, 2005; Emamjomeh and Sivakumar, 2009; Kabdaşlı *et al.*, 2012; Garcia-Segura *et al.*, 2017; Shahedi *et al.*, 2020), and others focus on particular metals (Barrera-Díaz *et al.*, 2012; Nidheesh and Singh, 2017). Separate reviews are available for capacitive deionization (Porada *et al.*, 2013; Chen *et al.*, 2020), electrodialysis (Arana Juve *et al.*, 2022; Cerrillo-Gonzalez *et al.*, 2024), and bioelectrochemical systems (Wang and Ren, 2014; Nancharaiah *et al.*, 2015). Wider reviews of heavy metal treatment compare electrochemical, adsorption, and membrane methods together (Bolisetty *et al.*, 2019; Shrestha *et al.*, 2021; Fei and Hu, 2023). Few of these studies, however, use energy efficiency as the main basis for comparison. Energy values are often given without stating whether they refer to the volume of water or the mass of metal, what the current efficiency was, or whether pumping and sludge handling were included. As a result, it is difficult to compare one technology fairly with another.

The present review was written to address this gap. Its objectives are:

(i) to collect, through a structured and targeted literature search, the evidence published between January 2000 and June 2026 on electrochemical removal and recovery of heavy metals;

(ii) to compare electrochemical processes with adsorption and membrane processes in terms of energy, residues and recovery potential;

(iii) to identify the design and operating factors that reduce energy use within each group of processes;

(iv) to assess solar-powered, hybrid and computer-optimised systems; and

(v) to bring together the available life cycle and cost studies and to suggest priorities for future research.

In this review, "energy-efficient" refers to the electrical energy required to reach a stated treatment target, expressed on a clearly defined basis. "Sustainable" is used in a wider sense that also covers the source of electricity, consumption of materials, residues, recovery of resources and cost.

REVIEW METHODOLOGY

This review is a critical, narrative review based on a structured but targeted search of the literature. It is not a full systematic review. The aim was to bring together the most relevant evidence on the energy use and sustainability of electrochemical methods, not to list every published study. The approach drew on the planning and reporting ideas used in systematic mapping in environmental science (Sordello *et al.*, 2020; Pullin *et al.*, 2022), but the formal screening and counting steps of a systematic review were not considered necessary and appropriate for this review.

Review Questions

The main question was whether there is evidence that electrochemical processes can remove or recover heavy metals from wastewater with lower energy use, or a smaller environmental footprint, than the reference technologies. Three further questions were also considered:

(i) which design and operating factors lower the energy demand in each group of processes;
(ii) how energy use is measured and reported; and
(iii) which systems have been tested on real effluents or at pilot scale.

Table 1: Keyword blocks used to guide the literature search.

Block	Search terms (combined with OR within a block)
A. Contaminant	"heavy metal*" OR arsenic OR cadmium OR chromium OR "Cr(VI)" OR copper OR lead OR mercury OR nickel OR zinc OR "toxic metal*"
B. Matrix	wastewater OR effluent OR "industrial water" OR groundwater OR "rinse water" OR "acid mine drainage"
C. Process	electrocoagulation OR electroflotation OR electrodeposition OR electrowinning OR electroreduction OR "capacitive deionization" OR electrosorption OR electrodialysis OR electrodeionization OR "microbial fuel cell" OR "microbial electrolysis cell" OR bioelectrochemical OR "electrochemical treatment"
D. Outcome	"energy consumption" OR "energy efficiency" OR "specific energy" OR "operating cost" OR sustainab* OR "life cycle assessment" OR "techno-economic" OR recovery OR solar OR photovoltaic OR "renewable energy"
Combination	A AND B AND C AND D; benchmark searches replaced block C with adsorption OR "reverse osmosis" OR nanofiltration OR ultrafiltration

Table 2: Criteria used to select documents for the review.

Criterion	Included	Excluded
Population (matrix)	Industrial, mining or municipal wastewater; contaminated groundwater; synthetic solutions mimicking these; soil-washing and leachate liquors	Solid-phase remediation without a liquid treatment step; drinking-water distribution corrosion studies
Exposure (intervention)	Electrochemical processes driven by an applied or biologically generated potential; hybrids in which the electrochemical step removes or recovers metals	Purely chemical, adsorptive or membrane processes (retained only as benchmarks)
Comparator	Alternative operating modes, electrode materials, power sources or benchmark technologies	None required for reviews
Outcomes	Metal removal or recovery efficiency together with at least one energy, cost, electrode-use, residue or life cycle indicator	Removal efficiency only, with no energy or sustainability information (except foundational mechanism papers)
Publication	January 2000 to June 2026; peer-reviewed articles, reviews, books, chapters, reports, proceedings; English	Pre-2000 items (except standards still in force); non-English; retracted items; predatory or unverifiable outlets

Sources and Search Terms

Only documents written in English and published between January 2000 and June 2026 were considered. The search was guided by the keyword blocks listed in Table 1, which combine terms for the contaminant, the water matrix, the process and the outcome of interest. Searches were run in Google Scholar and on the platforms of major publishers (ScienceDirect, SpringerLink, RSC, ACS, Wiley and MDPI). Google Scholar has very wide coverage of journals, books and grey literature (Martín-Martín *et al.*, 2018), but its results cannot be reproduced exactly, and it is not recommended as the only source for a systematic review (Gusenbauer and Haddaway, 2020). For this reason, the keyword searches were supplemented by backward citation chasing, in which the reference lists of key reviews were checked for further relevant studies. Books, book chapters, theses, conference papers and reports of bodies such as the World Health Organization (WHO) and the International Organization for Standardization (ISO) were also included where they provided basic data, definitions or standards.

Selection of Documents

Documents were selected by the author's judgement against the criteria given in Table 2. Priority was given to:

(i) reviews that summarise a process or a group of metals;
(ii) studies that explain removal mechanisms or the factors that control energy use;
(iii) studies that report energy consumption, operating cost, electrode consumption, residues or life cycle results;
(iv) studies on real effluents, continuous operation or pilot-scale plants; and
(v) recent work from 2020 to 2026 that introduces new materials, power supplies or process combinations.

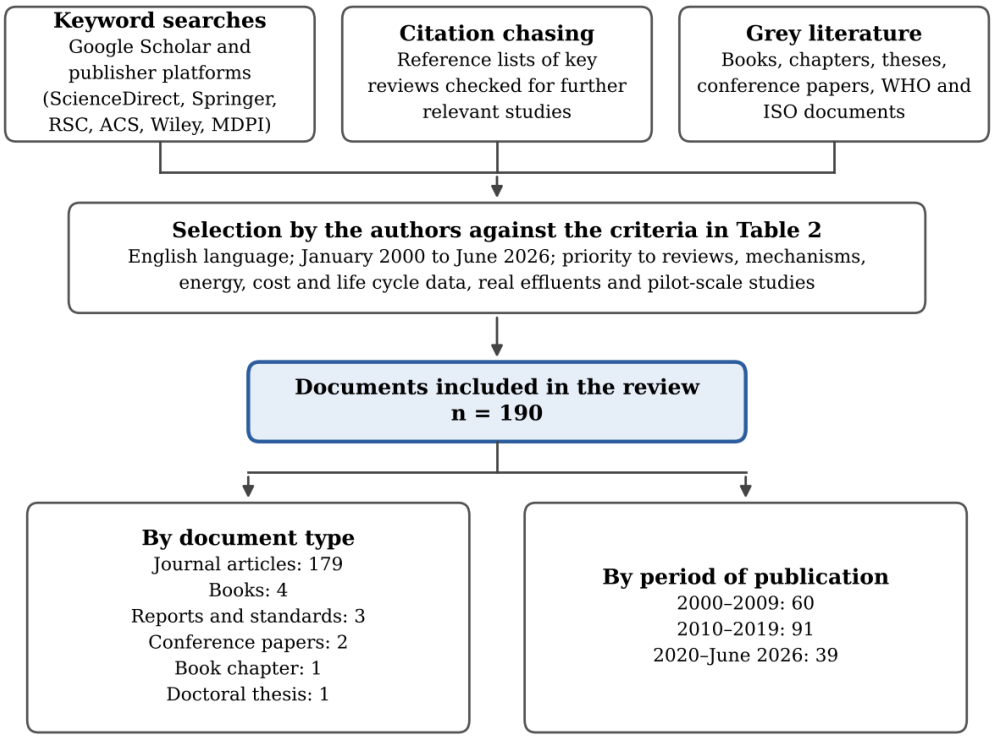


Fig. 1: Sources and composition of the 190 documents included in the review, by document type and by period of publication.

Older studies were kept where they remain the main reference for a mechanism or method. Formal steps used in systematic reviews, such as independent screening by two reviewers with an agreement statistic (Randolph, 2005) and the use of dedicated screening software (Ouzzani *et al.*, 2016), were not applied. The number of records found and excluded at each stage was therefore not recorded.

Composition of the Reviewed Literature

A total of 190 documents were included (Fig. 1). Of these, 179 are journal articles, 4 are books, 3 are reports or standards, 2 are conference papers, 1 is a book chapter and 1 is a doctoral thesis. By period, 60 were published between 2000 and 2009, 91 between 2010 and 2019, and 39 between 2020 and June 2026. The layout of Fig. 1 follows the general form of the flow diagrams recommended in ROSES (Haddaway *et al.*, 2018) and PRISMA 2020 (Page *et al.*, 2021), but it shows the sources and make-up of the reviewed literature instead of screening counts.

Data Extraction and Appraisal

The following information was extracted from each included study: the metal or metals studied; whether the water was synthetic or real; the scale and mode of operation (batch or continuous); the electrode materials and arrangement; the type of power supply (DC, AC, pulsed, solar or microbial); the current density or cell voltage; the removal or recovery efficiency; and any energy values, together with the basis on which they were reported. Each study was checked for how completely energy use had been accounted for, using the reporting framework described later. Energy values were used only when it was clear whether they referred to volume of water, mass of metal or a single operating cycle. Because the waters studied and the reporting methods varied so widely, no statistical pooling of results was attempted.

Limitations of the Approach

Because the search was targeted and the selection was based on judgement, this review does not claim to cover every published study, and some relevant work may have been missed. The emphasis on studies that report energy and sustainability data may also give more weight to electrocoagulation, which has the largest body of such data, than to newer processes. These limitations are common in critical reviews and were accepted in order to allow a deeper discussion of the evidence.

HEAVY METALS IN WASTEWATER: SOURCES, TOXICITY AND TREATMENT TARGETS

The treatment target has a direct effect on the energy required, and so it needs to be stated before different technologies are compared. Reducing copper from 500 mg/L to 1 mg/L in an electroplating rinse is a very different task from reducing arsenic from 0.3 mg/L to below 0.01 mg/L in groundwater, even though both may be reported as “99% removal”. Table 3 lists the main metals of concern, their principal industrial sources, and the drinking-water guideline values of the World Health Organization, which are often taken as the most stringent reference (WHO, 2022).

Three properties of these metals have a strong bearing on the energy needed for electrochemical treatment. The first is the chemical form in solution. Metals such as Cu^{2+} and Ni^{2+} are present as cations and can be either reduced at a cathode or precipitated as hydroxides. Chromate and arsenate, on the other hand, are anions, and they must be reduced, adsorbed onto flocs, or captured by electrodes that attract anions (Barrera-Díaz *et al.*, 2012; Nidheesh and Singh, 2017). The second is the reduction potential, which varies widely between metals. Copper and silver are deposited easily, whereas zinc, cadmium, and nickel are reduced at potentials close to those at which hydrogen is evolved, so part of the current is lost to hydrogen production (Chen and Lim, 2005; Hunsom *et al.*, 2005). The third is the composition of real effluents. These usually contain complexing agents such as EDTA, cyanide, or ammonia, and other ions such as Ca^{2+}, Na^{+}, Cl^{-}, and SO_4^{2-}, all of which can change how the metal is removed and how much energy is needed (Oztekin and Yazicigil, 2006; Pociecha and Lestan, 2010; Kim *et al.*, 2020b).

Discharge limits for industrial effluents vary from country to country, but for most metals they generally fall in the range

Table 3: Main heavy metals and metalloids found in wastewater, their sources, and WHO drinking-water guideline values.

Metal (dominant aqueous form)	Main wastewater sources	Key health concerns	WHO guideline value (mg/L)
Arsenic (As(III), As(V) oxyanions)	Mining, smelting, geogenic groundwater, wood preservation	Skin lesions, cancers, cardiovascular disease	0.01 (provisional)
Cadmium (Cd^{2+})	Ni–Cd batteries, plating, pigments, phosphate fertiliser	Kidney damage, bone disease, carcinogen	0.003
Chromium (Cr(VI) oxyanions, Cr^{3+})	Electroplating, tanneries, cooling towers, pigments	Cr(VI) is carcinogenic and mutagenic	0.05 (total Cr, provisional)
Copper (Cu^{2+})	Plating, printed circuit boards, mining, wire drawing	Gastrointestinal and liver effects	2
Lead (Pb^{2+})	Lead–acid batteries, mining, solder, paints	Neurotoxicity, especially in children	0.01 (provisional)
Mercury (Hg^{2+})	Chlor-alkali, gold mining, instruments	Kidney and neurological damage	0.006 (inorganic)
Nickel (Ni^{2+})	Plating, stainless steel, batteries	Allergic dermatitis, possible carcinogen	0.07
Zinc (Zn^{2+})	Galvanising, plating, rayon, mining	Low toxicity to humans; ecotoxic to aquatic life	No health-based value

of about 0.1–5 mg/L. For arsenic in groundwater, the target is close to 0.01 mg/L. As discussed in later sections, it is at these low concentrations that the rate at which metal ions reach the electrode, rather than the chemistry itself, controls the energy consumption of electrochemical processes.

REFERENCE TECHNOLOGIES: ADSORPTION AND MEMBRANE PROCESSES

Adsorption and membrane separation are the main non-electrochemical alternatives. Any claim that an electrochemical process is efficient therefore has to be judged against them. Table 4 gives a qualitative comparison of these reference technologies with the electrochemical processes covered in this review.

Adsorption

Adsorption on activated carbon, metal oxides, biosorbents and engineered nanomaterials is simple to operate and can bring metal concentrations below 1 mg/L (Babel and Kurniawan, 2003; Mohan and Pittman, 2006; 2007; Wang and Chen, 2009; Hua *et al.*, 2012). It uses little electricity directly, only for pumping and mixing. Its environmental costs lie elsewhere: the energy and chemicals used to make the adsorbent, the acid or alkali needed for regeneration, the gradual loss of capacity with repeated use, and the disposal of spent adsorbent or concentrated eluate (Burakov *et al.*, 2018; Qasem *et al.*, 2021). In water containing several metals, adsorbents are often not very selective, and most published data come from batch tests with a single metal. Since adsorption only concentrates the metal and does not convert it to a useful form, it is increasingly combined with an electrochemical step that strips the metal from the adsorbent or deposits it as metal. Examples include electro-assisted adsorption and combined electrocoagulation and adsorption (Issabayeva *et al.*, 2006; Yang *et al.*, 2021).

Membrane Processes

Pressure-driven membranes are effective at retaining divalent metal ions. Ultrafiltration first requires the metal to be bound to a polymer or to surfactant micelles, as in polymer-enhanced and micellar-enhanced ultrafiltration (Juang and Shiau, 2000; Landaburu-Aguirre *et al.*, 2009). Nanofiltration and reverse osmosis retain the hydrated ions directly (Ozaki *et al.*, 2002; Mohsen-Nia *et al.*, 2007; Abdullah *et al.*, 2019). The energy they use depends on the osmotic pressure of the feed and on the fraction of water recovered. For dilute industrial streams, low-pressure membranes use relatively little energy per cubic metre, and their sustainability is decided mainly by fouling, scaling and the handling of the reject stream (Shannon *et al.*, 2008; Elimelech and Phillip, 2011; Werber *et al.*, 2016). A bibliometric study has shown that newer membrane options for heavy metals, namely electrodialysis, membrane distillation and forward osmosis, have received much less attention than pressure-driven processes (Carmona and Abejón, 2023).

MEASURING ENERGY USE: A COMMON BASIS FOR COMPARISON

Energy values for electrochemical treatment of heavy metals can only be compared when the basis of calculation, the current efficiency, and the parts of the plant included are stated. Many of the papers reviewed give a single figure for "energy consumption" without this information. The definitions below are used in this review and are suggested for future studies.

The most widely reported figure is the specific energy consumption per unit volume, SECV, in kWh/m³ (Bayramoglu *et al.*, 2004; Kobya *et al.*, 2010):

$$\mathrm{SEC}_V = \frac{U I t}{1000\, V} \quad (1)$$

Table 4: Qualitative comparison of reference and electrochemical technologies for heavy metal wastewater (compiled from the reviews cited in the text).

Technology	Main removal mechanism	Main energy driver	Residue or secondary stream	Metal recovery potential	Best-suited feed
Adsorption	Surface complexation, ion exchange	Pumping; embodied energy of adsorbent and regenerants	Spent adsorbent or acidic eluate	Indirect, via elution	Dilute, low-volume polishing
Nanofiltration and reverse osmosis	Size and charge exclusion	Hydraulic pressure	Concentrate (10–25% of feed)	Indirect, concentrate needs further treatment	Dilute, low-fouling streams
Electrocoagulation	In-situ hydroxide coagulation, adsorption, co-precipitation, reduction	Cell voltage × charge; anode dissolution	Metal hydroxide sludge	Low (sludge) unless valorised	Mixed metals, moderate concentration
Electrodeposition and electroreduction	Cathodic reduction to metal or lower valence	Cell voltage; mass transfer; side reactions	Little or none	High (metal foil or powder)	Concentrated, single noble metals
Capacitive deionization and electrosorption	Double-layer or faradaic ion storage	Charging energy, partly recoverable	Brine on discharge	Moderate (concentrated stream)	Dilute, low-salinity streams
Electrodialysis and electrodeionization	Ion migration across ion-exchange membranes	Stack voltage; membrane resistance	Concentrate	Moderate to high	Rinse waters, acidic effluents
Bioelectrochemical systems	Biocatalysed anodic oxidation driving cathodic reduction	Little external energy; can produce power	Biomass; metal deposits on cathode	High for reducible metals	Effluents with biodegradable organics

where U is the average cell voltage (V), I the current (A), t the treatment time (hrs.) and V the volume of water treated (m³). When alternating or pulsed current is used, U and I should be root-mean-square or time-averaged values, not the peak or set values (Zhang *et al.*, 2021). SEC_V does not show how much metal was actually removed, so it should be reported along with the energy per unit mass of metal, SEC_m, in kWh per kg of metal removed:

$$\mathrm{SEC}_m = \frac{U\,I\,t}{V\,(C_0 - C_e)} \quad (2)$$

where C_0 and C_e are the initial and final metal concentrations (kg/m³). Faraday's law relates the electric charge passed to the mass of metal dissolved from a sacrificial anode or deposited on a cathode:

$$m = \frac{I\,t\,M}{z\,F} \quad (3)$$

where M is the molar mass, z is the number of electrons transferred, and F is the Faraday constant (96,485 C/mol). Current efficiency (CE) is the ratio of the mass actually dissolved or deposited to the mass predicted by Faraday's law. In electrocoagulation with aluminium, CE values above 100% are common, because the aluminium also dissolves chemically and by pitting (Mouedhen *et al.*, 2008; Mechelhoff *et al.*, 2013). In electrodeposition from dilute solutions, on the other hand, CE is often well below 50%, because much of the current is used to produce hydrogen (Chen and Lim, 2005).

Two simple calculations give an idea of the quantities involved. For copper deposition (z = 2, M = 63.55 g/mol), Faraday's law gives a charge requirement of about 844 Ah per kg of copper. At a cell voltage of 2.0 V and 100% CE, this corresponds to about 1.7 kWh/kg, and at 50% CE to about 3.4 kWh/kg. In electrocoagulation, 1 Ah dissolves about 0.34 g of aluminium (as Al^{3+}) or about 1.04 g of iron (as Fe^{2+}). This explains why iron anodes supply more coagulant for the same charge, although the floc formed from iron also depends on how quickly Fe(II) is oxidised (Lakshmanan *et al.*, 2009).

Operating cost is usually calculated as the sum of the cost of electricity and the cost of the electrode metal consumed (Bayramoglu *et al.*, 2004; Demirbas and Kobya, 2017):

$$\mathrm{OC} = a\,\mathrm{SEC}_V + b\,\mathrm{EC}_{\mathrm{electrode}} + c\,C_{\mathrm{chem}} \quad (4)$$

where a, b and c are the unit prices of electricity, electrode metal and chemicals, and $\mathrm{EC}_{\mathrm{electrode}}$ and C_{chem} are the amounts used per cubic metre. For capacitive and electrodialysis processes, energy is better expressed per mole of ions removed. Thermodynamic energy efficiency, defined as the minimum Gibbs energy of separation divided by the energy actually supplied, is also useful (Qin *et al.*, 2019). For capacitive cells, the energy recovered when the cell is discharged should be reported as well (Długołęcki and van der Wal, 2013).

On this basis, a complete energy report should include:

(i) SEC_V and SEC_m;
(ii) the current efficiency;
(iii) the type of power supply and how power was measured;
(iv) whether pumping, mixing, pH adjustment, and sludge dewatering are included; and
(v) the source of electricity, since this determines the carbon footprint.

Very few of the studies reviewed report all five.

ELECTROCOAGULATION

Electrocoagulation (EC) has been studied more than any other electrochemical method for heavy metal removal, and it has the most field experience. It is also the method in which a good deal of energy is often lost through ohmic resistance and electrode passivation that could be avoided. In EC, sacrificial aluminium or iron anodes release metal ions that hydrolyse to form hydroxide flocs. Dissolved heavy metals are removed by co-precipitation, adsorption and surface complexation on these flocs. Metals that can be reduced may also be removed at the cathode or by reaction with Fe(II). Hydrogen bubbles formed at the cathode can carry the flocs to the surface (Mollah *et al.*, 2001; 2004; Holt *et al.*, 2005; Emamjomeh and Sivakumar, 2009).

Removal of Individual Metals

Removal efficiencies above 95% have been reported for Cu, Zn, Ni, Cd, Pb, Mn and Cr in synthetic solutions and plating wastewaters using aluminium or iron electrodes (Adhoum *et al.*, 2004; Escobar *et al.*, 2006; Heidmann and Calmano, 2008; Shafaei *et al.*, 2010; Akbal and Camcı, 2011; Al Aji *et al.*, 2012). Real electroplating rinse waters, tannery effluents, and soil-washing solutions have also been treated with good results (Meunier *et al.*, 2006; Espinoza-Quiñones *et al.*, 2009; Kobya *et al.*, 2010; Pociecha and Lestan, 2010). Iron anodes are usually chosen for Cr(VI), because the Fe^{2+} released at the anode reduces chromate to Cr^{3+}, which then precipitates with the iron hydroxide (Gao *et al.*, 2005; Mouedhen *et al.*, 2009; Arroyo *et al.*, 2009; Bhatti *et al.*, 2009). For arsenic, iron-based EC oxidises As(III) and binds the resulting As(V) to Fe(III) oxides and hydroxides, and the structure of these solids has been studied in detail using synchrotron methods (Kumar *et al.*, 2004; Parga *et al.*, 2005; Gomes *et al.*, 2007; van Genuchten *et al.*, 2012). Several reviews have confirmed these findings (Bazrafshan *et al.*, 2015; Nidheesh and Singh, 2017; Khan *et al.*, 2023). The presence of complexing agents such as cyanide changes the main removal mechanism (Kim *et al.*, 2020b).

Factors That Control Energy Demand

The energy needed in EC depends on three things: the charge required to produce enough coagulant, the cell voltage at which this charge is supplied, and the treatment time. The charge required increases with the amount of metal and with other substances that use up coagulant, such as phosphate, silicate, and natural organic matter. Wan *et al.* (2011), for example, found that silicate and phosphate in groundwater considerably increased the dose of iron needed to remove arsenic. The cell voltage is made up of the equilibrium potential, the overpotentials at the two electrodes, the voltage drop across the solution, and the voltage drop across any passive film. The voltage drop across the solution increases with the electrode gap and decreases as conductivity rises. This is why adding chloride lowers energy use and also helps

to break down passive films, although it may lead to chlorinated by-products (Kobya *et al.*, 2003; Mouedhen *et al.*, 2008; Khan *et al.*, 2023).

Current density is the most important operating variable. A higher current density shortens the treatment time, but because of resistive losses the energy used rises roughly with the square of the current. Studies that have optimised EC with response surface methodology usually find an intermediate current density that gives the best balance between removal, energy use and electrode consumption (Bhatti *et al.*, 2009; Akbal and Camcı, 2011; Khan *et al.*, 2023). The rate at which charge is applied, in coulombs per litre per minute, is also important, separately from the total charge. Amrose *et al.* (2013) and Delaire *et al.* (2017) showed that lower dosing rates removed more arsenic per coulomb, because the Fe(II) had more time to oxidise. For anionic contaminants, therefore, slower operation at lower current can make better use of the energy supplied.

Electrodes, Passivation and Current Waveform

Aluminium anodes become covered with oxide films, and cathodes become fouled with hydroxide and carbonate deposits. Both effects raise the cell voltage over time, but they are seldom noticed in short batch tests (Mouedhen *et al.*, 2008; Mechelhoff *et al.*, 2013). Three remedies have been studied. Reversing the polarity of the electrodes at intervals spreads the wear between them and dissolves the cathode deposits (Garcia-Segura *et al.*, 2017). Alternating current and alternating pulse current (APC) limit the growth of passive films (Mao *et al.*, 2008; Yang *et al.*, 2015). Vasudevan *et al.* (2011) found that AC used less energy and less electrode material than DC for cadmium removal. Keshmirizadeh *et al.* (2011) obtained the same Cr(VI) removal with APC in a shorter time, which reduced the operating cost. Similar savings with AC and pulsed operation have been reported for wastewaters containing dyes, oil, and polymers (Eyvaz *et al.*, 2009; Bian *et al.*, 2019; Zhang *et al.*, 2021). These results suggest that the benefit is not limited to one type of water, but long-term comparisons on real heavy metal effluents are still few.

The way in which the electrodes are connected also affects energy use. In a monopolar parallel arrangement, the cell runs at low voltage and high current, while a bipolar series arrangement runs at higher voltage and lower current, and the total power for the same coagulant dose differs between the two (Kobya *et al.*, 2011; Al Aji *et al.*, 2012). Using aluminium and iron electrodes together can combine the higher coagulant yield of iron with the faster settling of aluminium flocs (Gomes *et al.*, 2007; Kobya *et al.*, 2011).

Reactor Design and Continuous Operation

Most EC studies are carried out in stirred beakers in batch mode. Such tests cannot show the effects of residence-time distribution, back-mixing, or gas hold-up. Studies with continuous reactors show that the use made of the coagulant depends strongly on the flow pattern in the reactor (Lu *et al.*, 2015). Models of EC reactors are still mostly empirical (Hakizimana *et al.*, 2017). Field-scale evidence for heavy metals comes mainly from arsenic removal. A community-scale iron EC plant in West Bengal, India, treating real groundwater, reported arsenic concentrations in the treated water below the limit of 10 µg/L (Amrose *et al.*, 2014). This is still one of the few long-term demonstrations for which operating data have been published. Broader reviews also note that pilot and full-scale EC plants remain uncommon (Kuokkanen *et al.*, 2013; Sahu *et al.*, 2014; Mao *et al.*, 2023).

Sludge Generation and Management

EC sludge is denser and easier to dewater than lime sludge, and less of it is usually produced than in chemical coagulation (Mollah *et al.*, 2004; Meunier *et al.*, 2006). It is, however, still a hazardous material containing heavy metals. If EC is to be considered sustainable, the sludge has to be put to use, for example as an adsorbent, a catalyst or a building material, or the metals in it have to be recovered. Recent life cycle work supports this view, as discussed later (Goyal and Mondal, 2026).

ELECTRODEPOSITION AND ELECTROCHEMICAL REDUCTION

Electrodeposition is the only method in this group that turns the dissolved metal directly into a saleable product without producing sludge. It works efficiently when the metal concentration is high, but its efficiency falls sharply in dilute solutions because the rate at which ions reach the electrode becomes limiting. Reduction of Cu^{2+}, Ag^{+}, Ni^{2+}, Zn^{2+} or Pb^{2+} to metal at a cathode has long been used in electrowinning and in recovering metals from plating baths (Jüttner *et al.*, 2000; Walsh, 2001; Chen, 2004). Reviews of electrochemical recovery group the available methods into electro-reduction, electro-oxidation, electro-precipitation, and electro-adsorption (Yang *et al.*, 2021).

Limitations At Low Metal Concentration

At low concentrations, the limiting current density is small, so a large part of the applied current is used to produce hydrogen and the current efficiency falls. Chen and Lim (2005) recovered precious metals efficiently from concentrated solutions but found that efficiency fell as the concentration decreased. Hunsom *et al.* (2005) modelled the reduction of copper from an industrial effluent and showed how current efficiency depends on concentration and flow rate. Complexing agents such as EDTA and cyanide shift the reduction potential to more negative values and reduce efficiency further (Oztekin and Yazicigil, 2006). In such cases, the complexes often have to be destroyed by oxidation before or during deposition (Anglada *et al.*, 2009; Chaplin, 2014; Sirés *et al.*, 2014).

Three-Dimensional and High-Surface-Area Electrodes

The usual engineering solution is to increase the electrode area per unit volume and improve mass transfer. Packed-bed, fluidised-bed, carbon felt, and reticulated vitreous carbon (RVC) cathodes have a much larger surface area per unit volume than flat plates. This allows a useful current to be passed at a low current density, and therefore at a lower overpotential (Zhang *et al.*, 2013; Walsh *et al.*, 2016; Arenas *et al.*, 2019). Issabayeva *et al.* (2006) deposited copper and lead on activated carbon made from palm shell in a flow-

through cell. Rodríguez-Valadez *et al.* (2005) reduced Cr(VI) to Cr(III) on RVC electrodes in a parallel-plate flow reactor. Electrodes modified with polyaniline have also been used for Cr(VI) reduction (Ruotolo and Gubulin, 2005; Barrera-Díaz *et al.*, 2012). For solutions containing several metals, cyclic electrowinning and precipitation can recover the metals one after another (Grimshaw *et al.*, 2011), and the metal deposited can be selected by controlling the electrode potential (Tran *et al.*, 2017).

New Electrode Materials and Current Waveforms

Since about 2017, alternating or rectified current waveforms have been used to overcome the problem of dilute solutions. Liu *et al.* (2017a) used half-wave rectified alternating current to extract uranium from seawater and obtained a higher capacity and a faster rate than with ordinary adsorption. The alternating field first drew the ions to the electrode and then deposited them as neutral compounds. Liu *et al.* (2019) used a similar DC/AC method with graphene oxide electrodes to remove and recover several heavy metal ions present at low concentration. Electrodes coated with redox-active polymers form a second group. Su *et al.* (2018) showed that a polyvinyl-ferrocene electrode could selectively capture chromium and arsenic anions even when other anions were present in much larger amounts. Kim *et al.* (2020a) used two different redox-polymer electrodes to capture As(III), oxidise it to the less toxic As(V), and then release it, all in one process. In these methods, the electrode potential is used to switch the binding of the target ion on and off, so the acid and alkali normally needed to regenerate an adsorbent are no longer required (Su and Hatton, 2017; Su, 2020).

The energy picture for these methods is not yet clear. They avoid regeneration chemicals and can work at low cell voltages. However, most have been tested only in small cells with clean electrolytes, and it is not yet known how stable the redox polymers and graphene oxide will be over long periods in real effluents.

CAPACITIVE DEIONIZATION AND ELECTROSORPTION

Capacitive deionization (CDI) is energy-efficient for heavy metals only under fairly narrow conditions, namely in dilute, low-salinity water where the metals make up only a small part of the total dissolved solids. In more saline water, most of the charge is used to remove other ions. In CDI, a low cell voltage, usually below about 1.2 V to avoid splitting water, stores ions in the electrical double layers of porous carbon electrodes. When the cell is discharged, the ions are released into a smaller volume of water, and part of the stored energy can be recovered (Oren, 2008; Porada *et al.*, 2013; Suss *et al.*, 2015).

Electrosorption of Heavy Metals

Early work on carbon aerogel electrodes showed that ions with higher charge are removed more readily (Gabelich *et al.*, 2002). Huang *et al.* (2016) compared the removal of lead, cadmium, and chromium on their own and in mixtures and found that the charge and hydrated size of the ions determined which were removed first. Fan *et al.* (2016) removed both As(III) and As(V), with the charged As(V) being removed more effectively. In general, selectivity between ions in CDI depends on their charge, their hydrated radius and the pore size of the electrode (Hou and Huang, 2013). Recent reviews show a clear move away from plain carbon towards modified carbons and towards faradaic, redox-active materials. These materials bind particular metals chemically instead of storing all ions by electrostatic attraction (Chen *et al.*, 2020; Srimuk *et al.*, 2020; Xu *et al.*, 2023). Graphene-based electrodes have also been studied because of their high conductivity and large accessible surface area (Li *et al.*, 2010; Liu *et al.*, 2017b).

Energy Performance and Its Limits

The energy advantage of CDI is often overstated. Comparisons with reverse osmosis show that CDI is competitive mainly for feeds of low salinity, and that its thermodynamic efficiency is usually low (Zhao *et al.*, 2013; Qin *et al.*, 2019). Several design choices improve it. In membrane CDI, ion-exchange membranes are placed in front of the electrodes to stop ions of the same charge from being expelled, which improves charge efficiency (Biesheuvel and van der Wal, 2010). Operation at constant current gives a more stable product water and uses less energy than operation at constant voltage (Jande and Kim, 2013). Recovering the energy stored during charging, which can be a significant part of the input, reduces net consumption further (Długołęcki and van der Wal, 2013). Battery-type and hybrid CDI electrodes store ions by intercalation or chemical conversion and so have a higher capacity (Pasta *et al.*, 2012; Lee *et al.*, 2014). Side reactions such as oxidation of the carbon and reduction of dissolved oxygen lower efficiency and shorten electrode life. In some cases, they can be put to use, for example, to reduce Cr(VI) at the cathode (Tang *et al.*, 2017; Zhang *et al.*, 2018).

Flow-Electrode CDI For Continuous Recovery

In flow-electrode CDI (FCDI), the electrodes are a carbon slurry that flows through the cell, so that water can be treated and metals concentrated continuously without stopping to discharge the cell. Recent studies have applied FCDI to nickel and to industrial wastewater containing several metals, and have optimised the voltage, pH, and slurry composition using response surface methodology (Pelarti and Mirbagheri, 2022; Elawadly *et al.*, 2025). Reviews of CDI for wastewater treatment identify heavy metal removal and energy recovery as active research areas (Bao *et al.*, 2023; Sayed *et al.*, 2023). An important question that remains open is whether the metal-loaded slurry can be regenerated, and the metal recovered, using less energy than the electrosorption step itself.

ELECTRODIALYSIS AND ELECTRODEIONIZATION

Electrodialysis (ED) is the most developed of the membrane-based electrochemical methods for heavy metals. Its main advantage is that it produces a concentrated metal solution that can be reused, instead of a sludge. In ED, an electric field drives cations and anions through alternating cation-exchange and anion-exchange membranes, producing a dilute stream and a concentrated stream. The energy used depends

on the quantity of ions transferred and on the electrical resistance of the stack (Strathmann, 2010; Campione *et al.*, 2018). ED is therefore efficient for moderately concentrated streams such as plating rinse waters and acidic metallurgical effluents. It is much less efficient for polishing very dilute water, because the dilute compartment then has a high resistance.

Removal And Recovery of Metals

Modelling and laboratory studies have shown how voltage, flow rate, temperature, and concentration affect metal removal by ED (Mohammadi *et al.*, 2005), and neural network models have been used to predict lead removal (Sadrzadeh *et al.*, 2009). Recent reviews describe the use of ED to concentrate and separate Cu, Ni, Zn, Cr, and other metals from acidic effluents, in some cases with bipolar membranes to regenerate acid and alkali (Arana Juve *et al.*, 2022; Cerrillo-Gonzalez *et al.*, 2024). Separating one metal from another, or metals from protons, depends on the use of monovalent-selective or chelating cation-exchange membranes and on keeping the current density below its limiting value (Tekinalp *et al.*, 2023).

Electrodeionization For Dilute Streams

In electrodeionization (EDI), the dilute compartment is filled with ion-exchange resin. The resin carries current even in very dilute water and is continuously regenerated by water splitting at the boundaries between resin and membrane. EDI therefore extends the electrodialysis principle to low concentrations at which ED alone becomes too resistive (Alvarado and Chen, 2014). Dermentzis (2010) combined electrostatic shielding with EDI to remove nickel from electroplating rinse water and obtained a concentrate that could be reused. EDI is attractive for closing the rinse-water loop in plating shops, where both water and metal can be recovered.

The main obstacles are the cost of the membranes, fouling and scaling by metal hydroxides where the local pH is high, and disposal of the concentrate. A bibliometric study has shown that ED for heavy metals has received relatively little attention compared with pressure-driven membranes, so there is still much scope for development (Carmona and Abejón, 2023).

BIOELECTROCHEMICAL SYSTEMS

Bioelectrochemical systems (BES) are the only electrochemical processes in this review that can remove metals while producing, instead of using, electrical energy. Their weakness is that they are slow, which limits the volume of water they can treat. In a microbial fuel cell (MFC), bacteria at the anode oxidise organic matter and pass electrons through an external circuit to the cathode (Rabaey and Verstraete, 2005; Logan *et al.*, 2006; Logan, 2008). Metal ions whose standard potentials are higher than those of the anode reaction, such as Cr(VI), Cu(II), Ag(I), and Au(III), can accept these electrons at the cathode and are reduced without any external power. Metals with more negative potentials, such as Ni, Cd, Pb, and Zn, need a small external voltage, applied in a microbial electrolysis cell (MEC). This voltage is still lower than in ordinary electrolysis, because the anode reaction is driven by the bacteria (Logan and Rabaey, 2012; Wang and Ren, 2014).

Metal Reduction in Microbial Fuel Cells

In an early study, Wang *et al.* (2008) showed that Cr(VI) could be reduced at the cathode while the cell was producing electricity. Li *et al.* (2008) later did the same with real electroplating wastewater. Biocathodes, on which microorganisms catalyse the reduction of the metal, lowered the cathode overpotential and improved both Cr(VI) reduction and power output (Huang *et al.*, 2010; Wang *et al.*, 2023). Copper has been recovered as metal while generating power (ter Heijne *et al.*, 2010; Tao *et al.*, 2011), and silver has been recovered in a similar way (Choi and Cui, 2012).

Sequential Recovery Using MFC and MEC Operation

Modin *et al.* (2012) first recovered copper in MFC mode and then recovered lead, cadmium and zinc in turn by applying increasing external voltages. This shows how the differences in reduction potential between metals can be used to separate them one by one with little energy. Reviews of MECs describe the cell designs used and the voltage losses involved in removing Co, Ni, Cu, Zn, Pb and Cd (Wu *et al.*, 2023). Wider reviews conclude that metals are removed in BES by direct bioelectrochemical reduction, by biological precipitation, and by adsorption on biomass (Nancharaiah *et al.*, 2015; Bagchi and Behera, 2020; Wang and He, 2020).

Limitations and Outlook

Current densities in BES are usually ten to a hundred times lower than in ordinary electrolysis, so treatment times are long and reactors are large. Other limitations are the toxicity of metals to the bacteria at the anode, the cost of membranes, and the low value of mixed deposits of metal and biomass. Recent reviews of BES for recovering metals from acid mine drainage, electronic waste, and smelter effluents report progress in combining BES with other technologies, but cost and slow reaction rates remain the main barriers to scale-up (Pines Pozo *et al.*, 2025). BES are also being studied for cleaning metal-contaminated sediments without any external power supply (Liu *et al.*, 2024). The most practical use of BES is likely to be the final treatment of effluents that contain both biodegradable organic matter and reducible metals, such as some food-processing and mixed industrial wastewaters. In such cases, the organic matter supplies the electrons that would otherwise have to come from the grid.

SOLAR-POWERED AND HYBRID SYSTEMS

The carbon footprint of an electrochemical process depends more on where its electricity comes from than on how many kilowatt-hours it uses. Running these processes on renewable power, and combining processes so that each one works under the conditions that suit it best, are the two most effective ways found in this review of making them more sustainable.

Direct Coupling with Solar Panels

Electrochemical cells run on low-voltage direct current, which makes them easy to connect to photovoltaic (PV) panels. Valero *et al.* (2008) showed that an EC reactor could

be connected directly to PV panels without batteries, with the flow rate adjusted to match the current available. Ortiz *et al.* (2006) modelled a PV-powered ED system for desalination. For heavy metals, Dermentzis *et al.* (2014) treated electroplating effluent containing Cr, Ni, and Cu with iron EC powered directly by PV panels without batteries. They achieved over 99% removal at about 15 kWh/m³, which is a high figure explained by the concentrated feed and the high current density used. Ali *et al.* (2021) removed nickel with aluminium electrodes powered by a solar cell, and Hussin (2018) developed solar-powered EC for lead removal using new electrode materials. The use of renewable energy for ED has been reviewed more generally by Mir and Bicer (2021). Two points come out of these studies. Changes in sunlight have to be matched by changing the flow rate or by short-term energy storage. Operating without batteries also avoids the environmental burden of making and disposing of lead–acid or lithium batteries.

The life cycle benefit of solar power has now been measured. In a scaled-up EC system with aluminium electrodes for arsenic and fluoride removal, the use of solar electricity reduced the global warming potential by about 37%. The capital cost increased by about 64%, but the operating cost fell by about 11%, and the plant remained profitable (Goyal and Mondal, 2026).

Hybrid Process Trains

Hybrid systems save energy when each unit treats the concentration range it handles best. Combinations reported in the literature include:

- EC with electroflotation, in which hydrogen from the cathode floats the flocs, so that no settling tank or filter is needed (Gao *et al.*, 2005; Merzouk *et al.*, 2009).
- EC as a pretreatment ahead of ultrafiltration or other membranes, which reduces fouling (Timmes *et al.*, 2010).
- EC followed by adsorption, which allows EC to run at a lower current density because the adsorbent removes the remaining metal (Hussin, 2018).
- Adsorption or ion exchange to capture the metal, followed by electrochemical elution and deposition to recover it (Issabayeva *et al.*, 2006; Yang *et al.*, 2021).
- CDI combined with membranes, ion exchange, or microbial desalination (Chen *et al.*, 2020; Sayed *et al.*, 2023).
- ED or EDI to concentrate rinse waters, followed by electrowinning of the concentrate (Dermentzis, 2010; Arana Juve *et al.*, 2022).

The last of these combinations is particularly promising. ED handles the dilute stream, in which direct deposition would be inefficient, while electrowinning is applied to the concentrate, where current efficiency is high. Energy is therefore used where each process works best.

COMPARISON OF TECHNOLOGIES ON ENERGY AND SUSTAINABILITY

No single electrochemical process is the most energy-efficient for every type of heavy metal wastewater. Each one works best within a certain range of concentration, set mainly by mass transfer in the case of deposition, by the total ion load in the case of electrodialysis and CDI, and by the coagulant demand in the case of EC (Fig. 2).

Deposition is best suited to concentrated solutions, while capacitive deionization, electrodeionization and redox-based capture suit dilute ones. EC, ED and BES occupy the middle range.

Table 5 shows the basis on which energy should be reported for each group of processes, together with typical magnitudes. The values are given as ranges or as calculated

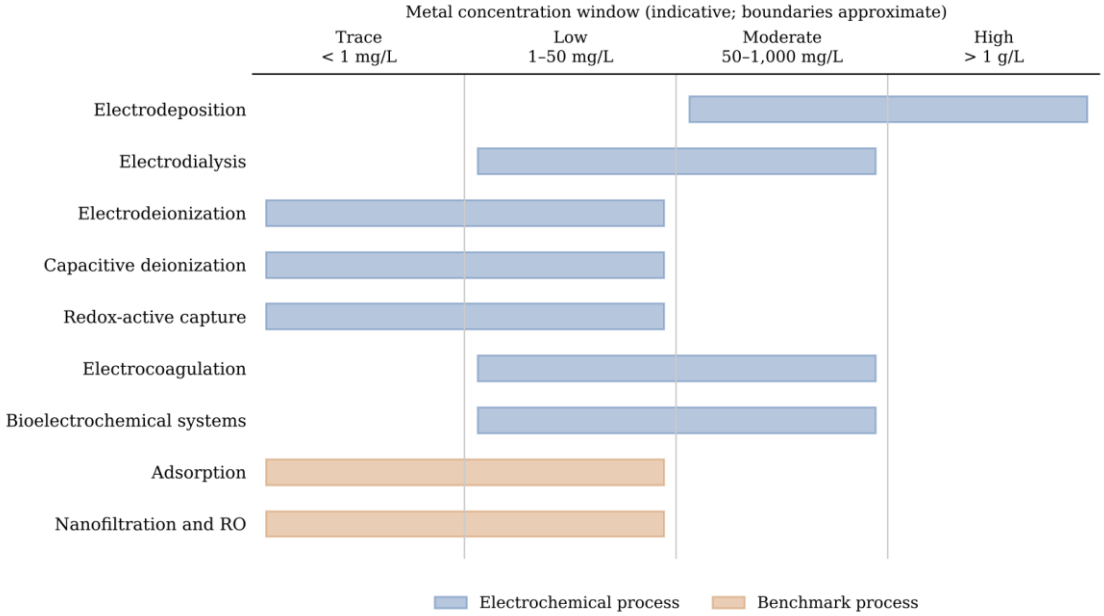


Fig. 2: Indicative concentration windows in which each process is energy-favoured, synthesised from the reviewed literature; boundaries are approximate.

Table 5: Basis for reporting energy and typical magnitudes for heavy metal treatment technologies.

Technology	Preferred energy basis	Indicative magnitude	Dominant determinants	Key sources
Electrocoagulation	kWh/m³ and kWh/kg metal	Usually below 1 to several kWh/m³; about 15 kWh/m³ reported for concentrated plating effluent	Current density, gap, conductivity, passivation, coagulant demand	Bayramoglu *et al.*, 2004; Kobya *et al.*, 2010; Dermentzis *et al.*, 2014; Khan *et al.*, 2023
Electrodeposition	kWh/kg metal	About 1.7 kWh/kg Cu at 2 V and 100% CE (derived in this review); rises steeply as CE falls	Concentration, mass transfer, hydrogen evolution, complexants	Chen and Lim, 2005; Hunsom *et al.*, 2005
Capacitive deionization	kWh/m³ or J per mol ions; energy recovered	Low for dilute, low-salinity feeds; part of charging energy recoverable	Salinity, charge efficiency, faradaic losses	Zhao *et al.*, 2013; Długołęcki and van der Wal, 2013; Qin *et al.*, 2019
Electrodialysis and EDI	kWh/m³ and kWh/kg metal	Scales with ions removed and stack resistance	Feed concentration, membrane resistance, limiting current	Strathmann, 2010; Campione *et al.*, 2018
Bioelectrochemical systems	Net W/m³ or kWh/kg metal	Net producer in MFC mode; small applied voltage in MEC mode	Anode kinetics, metal redox potential, internal resistance	Wang and Ren, 2014; Modin *et al.*, 2012
Adsorption (benchmark)	Embodied MJ per kg adsorbent and regenerant	Low direct electricity; burden in synthesis and regeneration	Capacity, cycles, regenerant use	Burakov *et al.*, 2018
NF and RO (benchmark)	kWh/m³	Set by osmotic pressure and recovery	Salinity, fouling, recovery ratio	Elimelech and Phillip, 2011; Werber *et al.*, 2016

benchmarks because published figures differ in their basis and completeness.

Energy is only one part of sustainability. Table 6 gives a qualitative assessment of each technology against six criteria, based on the evidence reviewed. The ratings represent the authors' judgement and are meant to help in choosing a technology, not to rank the processes in absolute terms.

PROCESS OPTIMISATION, LIFE CYCLE AND COST STUDIES

Optimisation and Process Control

Since the energy used in electrochemical treatment depends on several variables that interact with one another, multi-objective optimisation can bring useful savings. Response surface methodology has often been used to find the best balance between removal, energy use and electrode consumption (Bhatti *et al.*, 2009; Khan *et al.*, 2023). Artificial neural networks were applied to ED some years ago (Sadrzadeh *et al.*, 2009). More recently, Jain *et al.* (2022) combined an artificial neural network with a multi-objective optimisation algorithm to produce trade-off curves between removal efficiency and energy use for lead, cobalt and manganese removal by EC. Their results showed clearly how much removal has to be given up for a given saving in energy. Machine learning models can now predict EC performance for other pollutants with good accuracy (Albaba *et al.*, 2025). However, such models are only as reliable as the data used to train them, and most of these data come from short batch tests with synthetic solutions. Real-time control, in which conductivity, pH, and cell voltage are measured continuously, and the current is adjusted to supply only the charge needed, has hardly been tried for heavy metals. It may well be the largest untapped source of energy savings.

Table 6: Qualitative sustainability assessment (H = favourable, M = intermediate, L = unfavourable).

Criterion	EC	Electrodeposition	CDI and electrosorption	ED and EDI	BES	Adsorption	NF and RO
Direct energy demand	M	M (H at high concentration)	H at low salinity	M	H	H	M
Chemical input	H	H	H	H	H	L (regenerants)	M (antiscalants, cleaning)
Residue burden	L (sludge)	H	M (concentrate)	M (concentrate)	M	L (spent media or eluate)	L (concentrate)
Metal recovery potential	L	H	M	H	M	M	L
Renewable-power compatibility	H	H	H	H	Inherent	Not applicable	M
Maturity for heavy metals	M (field arsenic plants)	H (concentrated streams)	L	M	L	H	H

Life Cycle Assessment

Life cycle assessment (LCA) according to ISO 14040 compares processes over their whole life cycle (ISO, 2006). It has been widely applied to wastewater treatment in general but only rarely to electrochemical removal of heavy metals (Corominas *et al.*, 2013). The few studies available point in the same direction. For EC of municipal wastewater, electricity was the main contributor to impacts such as terrestrial ecotoxicity, and reducing the current density lowered the impacts with little loss in performance (Leovac Maćerak *et al.*, 2024). For arsenic and fluoride removal by EC with aluminium electrodes, Goyal and Mondal (2026) found that scaling up from a 1.7 L laboratory cell to a 650 L system roughly halved the global warming potential per unit of water treated, and that solar electricity reduced it further. Their analysis also suggested an overall benefit to human health from the treated water. These results show that LCAs based only on laboratory-scale data can overestimate impacts, and that the source of electricity and the scale of the plant need to be modelled explicitly. The principles of green engineering, such as reducing energy and material use and designing for recovery, provide a useful framework for such studies (Anastas and Zimmerman, 2003).

Cost Assessment

Operating cost models for EC, based on electricity and electrode consumption, have been in use for about two decades (Bayramoglu *et al.*, 2004). Studies comparing EC with chemical coagulation report similar or lower costs for EC, especially when the cost of sludge handling is included (Demirbas and Kobya, 2017). Full techno-economic assessments, including capital cost, discounting, and income from recovered products, are rare. Goyal and Mondal (2026) provide a recent example. In their best case, with a three-shift operation and use of the sludge, the project had a positive net present value, an internal rate of return of about 49% and a payback period of about two years. Similar analyses for electrodeposition, CDI, ED, and BES treating heavy metal wastewater are largely missing.

Resource Recovery

The emphasis in this field is moving from removal of metals to their recovery. Electrochemical methods are important here because they can produce metals as deposits, as concentrated salt solutions, or as regenerated acids (Yang *et al.*, 2021; Arana Juve *et al.*, 2022). Electrochemically controlled separations allow selective binding and release of both valuable and toxic metals without chemicals (Su, 2020; Srimuk *et al.*, 2020). Bioelectrochemical recovery from mine drainage and electronic waste is advancing, although not yet at commercial scale (Pines Pozo *et al.*, 2025). For EC, recovery depends on finding a use for the sludge, and evidence on this is limited. The purity and market value of recovered products, and whether regulators treat them as products or as wastes, are seldom reported.

STRENGTHS AND WEAKNESSES OF THE EVIDENCE

The literature shows that electrochemical treatment of heavy metals is effective. It does not yet support most of the claims made about energy efficiency, because of five weaknesses that appear again and again.

First, the test solutions are often unrealistic. Most studies use synthetic solutions containing a single metal, with NaCl or Na_2SO_4 added to increase conductivity. This hides the effect of competing ions, organic matter, complexing agents, and scaling on the charge needed and on electrode life (Wan *et al.*, 2011; Kim *et al.*, 2020b).

Second, experiments are short and carried out in batch mode. Passivation, fouling, and electrode wear develop over hours or weeks, whereas most experiments last from a few minutes to a few hours. Energy use measured with fresh electrodes is therefore lower than would be expected in long-term operation (Mechelhoff *et al.*, 2013; Hakizimana *et al.*, 2017).

Table 7: Priority research gaps and suggested actions.

Gap	Current evidence	Recommended action	Priority
Inconsistent energy reporting	Volumetric energy only; basis often unstated	Report SEC_V, SEC_m, current efficiency, waveform-integrated power, ancillary loads and electricity source	High
Synthetic matrices	Single-metal solutions dominate	Test on real effluents with full ionic and organic characterisation	High
Long-term stability	Hours-long batch runs	Run continuous trials of at least several weeks, tracking cell voltage, passivation and electrode loss	High
Residue fate and valorisation	Sludge and concentrates rarely characterised	Leaching tests; metal recovery or reuse of EC sludge; electrowinning of ED and CDI concentrates	High
Life cycle and cost evidence	Few LCAs and TEAs, mostly for EC	Scale-explicit LCA and TEA for every process family, including revenue from recovered metal	High
Renewable integration	Small PV-EC and PV-ED demonstrations	Variable-load operating strategies, battery-free control, hybrid solar–grid supply	Medium
Real-time control	Offline RSM and ANN models	Sensor-based closed-loop dosing of current and charge, with digital twins	Medium
Selective capture materials	Redox polymers and faradaic CDI in clean electrolytes	Durability and selectivity tests in real multi-metal effluents	Medium
BES throughput	Low current density, long residence time	Biocathode engineering; coupling BES with abiotic polishing steps	Medium
Underrepresented metals and regions	Focus on Cu, Cr, Ni, As; few studies on Hg or Tl; limited low-income field data	Broaden target metals; decentralised trials in high-burden regions	Medium

Third, energy accounting is incomplete. Few studies report energy per unit mass of metal, current efficiency or the energy used by pumps and other auxiliary equipment. With pulsed and alternating current, power is sometimes calculated from set values rather than from the actual waveform. The cell voltage is rarely divided into its thermodynamic, kinetic and resistive parts, although doing so would show where the losses occur (Bard and Faulkner, 2001).

Fourth, residues are seldom followed up. EC sludge, CDI, ED concentrates and spent redox electrodes are rarely tested for leaching or for recovery value. Leaving them out makes electrochemical methods appear cleaner than they are.

Fifth, there is little evidence from larger-scale operations. Apart from community arsenic plants and established electrowinning practice, continuous pilot data on real heavy metal effluents are scarce for all five groups of processes (Amrose *et al.*, 2014; Mao *et al.*, 2023).

RESEARCH GAPS AND FUTURE DIRECTIONS

Table 7 turns these weaknesses into a list of research needs, ordered by their likely effect on energy efficiency and sustainability.

CONCLUSIONS

This review has examined the literature published between January 2000 and June 2026 on energy-efficient and sustainable electrochemical methods for removing and recovering heavy metals from wastewater. The main conclusions are as follows.

- No single electrochemical process is the most energy-efficient in all cases. Electrodeposition is best for concentrated streams. Electrodialysis, electrocoagulation and bioelectrochemical systems suit intermediate concentrations, and electrodeionization, capacitive deionization and redox-based capture suit dilute streams. Choosing the process according to the concentration range, often within a hybrid system, is the most effective single way of saving energy.
- In electrocoagulation, energy use can be reduced by working at a moderate current density, increasing conductivity where this is acceptable, reducing the electrode gap, and limiting passivation through alternating or pulsed current and polarity reversal. For anionic contaminants such as arsenic, slower charge dosing removes more contaminant per coulomb.
- Three-dimensional electrodes, deposition with alternating or rectified current, and electrodes with redox-active coatings make it possible to recover metals from dilute streams without chemical regeneration. These approaches still need to be tested in real effluents over long periods.
- Bioelectrochemical systems can reduce Cr(VI), Cu, Ag, and other easily reduced metals while generating electricity. Their use is limited by low treatment rates rather than by energy.
- The source of electricity has the greatest effect on life cycle impacts. Both direct solar coupling and larger plant size reduce the environmental footprint, and recent cost studies show that solar-powered electrocoagulation with use of the sludge can be profitable.
- The evidence base is weakened by the use of synthetic solutions, short batch tests, incomplete energy accounting, and neglect of residues. A common reporting framework, such as the one proposed here, is needed before technologies can be compared reliably.

Electrochemical treatment can become a low-carbon option that supports metal recovery, but only if researchers move from reporting maximum removal under ideal conditions to reporting complete energy and residue data from continuous operation on real effluents.

ORCID iD

Surinder Deswal: 0000-0001-7256-142X

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Conflict of Interest

The author declares that there is no conflict of interest regarding the publication of this manuscript. In addition, the ethical issues, including plagiarism, informed consent, misconduct, data fabrication and/ or falsification, double publication and/or submission, and redundancy, have been completely observed by the author.

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