Civil and Sustainable
Urban Engineering

Review
Volume 5(1), 2025, 30–52
https://doi.org/10.53623/csue.v5i1.629

Sustainable Water Management Strategies for Mitigating Pesticide Pollution in Urban and Agricultural Areas

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SUBMITTED: 8 March 2025; REVISED: 30 April 2025; ACCEPTED: 9 May 2025

ABSTRACT: The indiscriminate use of pesticides in Malaysian agriculture poses serious risks to both human health and groundwater quality.This study aims to evaluate the extent of pesticide contamination in Malaysian groundwater, identify its major sources, and examine current mitigation efforts. The primary routes of contamination include direct application, soil leaching, and surface runoff, with over twenty pesticide compounds listed as priority hazardous substances, commonly linked to oil palm, rice, and vegetable farming. Residential and industrial activities also contribute to the pollutant load. Due to their long environmental persistence, pesticides threaten aquatic ecosystems through bioaccumulation and biomagnification and increase the risk of severe health issues, including neurological disorders, reproductive problems, and cancer. Regulatory controls such as exposure limits and monitoring programs have been implemented to manage these risks. This review concludes that while regulatory mechanisms exist, more robust and proactive approaches are needed to mitigate groundwater contamination. Future efforts should focus on expanding the adoption of sustainable farming practices, strengthening groundwater monitoring, and enhancing regulatory enforcement to ensure long-term environmental and public health protection.

KEYWORDS: Pesticides; groundwater; Malaysia; agriculture; environmental health; public health

1. Introduction

Malaysia, a nation widely known for its scenic landscapes and expanding agricultural sector, has faced a critical environmental challenge beneath the surface, groundwater contamination from pesticides [1]. As a result of unintentional seepage into groundwater reservoirs, pesticides, although essential to modern agriculture's pursuit of higher yields, have posed significant threats to both the environment and public health [1]. Understanding the causes and consequences of pesticide contamination in Malaysian groundwater has been essential for developing effective mitigation strategies. The use of pesticides was widespread in Malaysian agriculture due to the need to increase crop yields and meet the demands of a growing population [2]. In comparison, the European Union (EU), through regulatory bodies such as the European Chemicals Agency (ECHA) and the European Food Safety Authority (EFSA), imposed strict controls on pesticide use. Several pesticides considered harmful to both the environment and public health have been banned or severely restricted in the EU [3]. Additionally, the EU adopted the precautionary principle, which allowed for the restriction or prohibition of pesticides with uncertain safety until they were proven to be safe [3]. However, in Malaysia, the careless and mismanaged use of pesticide agents enabled them to infiltrate groundwater sources, thereby compromising the quality of this vital resource. Since many regions in Malaysia relied primarily on groundwater for drinking water, the presence of pesticide residues raised serious concerns about long-term human exposure and associated health risks [3]. This study aimed to review the issue of pesticides in Malaysian groundwater by investigating their sources, pathways, and environmental and public health impacts. It also examined current legal frameworks and mitigation strategies, while exploring opportunities to promote sustainable agricultural practices that reduce reliance on chemical pesticides and minimize groundwater contamination. Furthermore, the review underscored the urgent need to address the environmental and health implications of pesticide pollution in Malaysian groundwater. Malaysia can safeguard its groundwater resources, protect public health, and ensure the sustainability of its agricultural sector only through coordinated action, supported by comprehensive research and multi-stakeholder collaboration [4]. This research examined the sources, distribution, and migration of pesticides; the application of mathematical models in groundwater systems; the environmental evolution of pesticides; their effects on living organisms and community health; and available remediation technologies to manage pesticide transport in groundwater.

2. Sources, Distribution and Movement of Pesticides

2.1. Point sources.

Pesticides were introduced or released directly into the groundwater system in Malaysia. Point sources, typically located in agricultural regions, contributed to pesticide pollution and often had a more concentrated and localized impact compared to diffuse sources [5]. Pesticides sprayed on agricultural land had the potential to seep into the soil and eventually infiltrate groundwater. They were also washed away by rainfall or irrigation. The Cameron Highlands in Pahang and the Muda Agricultural Development Authority (MADA) area in Kedah were notable examples of intensive farming regions most affected by runoff and leaching [6]. Potential point sources of contamination included facilities that handled and stored pesticides and other chemicals. Improper storage practices often resulted in leaks or spills, releasing chemicals directly into the soil and groundwater [7]. Groundwater pollution also arose from accidental discharges during pesticide transportation [7]. Additionally, chemical manufacturing facilities producing pesticides contributed to contamination through improper disposal of waste or wastewater containing chemical residues [8]. In some cases, treated and untreated industrial discharges from chemical storage areas allowed pesticides to seep into groundwater [8]. Without proper containment measures, pesticide containers, residues, or contaminated soil from agricultural activities were sometimes disposed of in landfills [9]. Over time, aquifers became contaminated as leachate containing dissolved pesticides seeped into surrounding soil and groundwater [9]. Furthermore, groundwater pollution also resulted from the illegal disposal of pesticide containers or waste in both urban and rural areas [10]. In regions with weak regulatory enforcement or inadequate disposal infrastructure, individuals resorted to unlawful pesticide disposal, leading to localized clusters of contamination. These illegal disposal sites posed significant threats to both human health and groundwater quality.

2.2. Non-point sources.

Non-point sources referred to the diffuse or widespread sources of contamination that resulted in the presence of pesticides in Malaysian groundwater, without originating from a single, identifiable location. Unlike point sources, which discharged pesticides directly into groundwater at specific sites, non-point sources distributed contaminants over a much larger area, making it difficult to pinpoint the exact source of pollution [5]. One of the major non-point sources of pesticides in groundwater was runoff from agricultural lands where chemicals were applied to crops [11]. Pesticides from treated fields were often carried by rainfall or irrigation water, which washed them off the soil surface into nearby water bodies such as rivers, streams, and eventually into groundwater. Due to its wide distribution, agricultural runoff played a significant role in pesticide contamination of groundwater in Malaysia [11]. A notable example was the MADA area in the northern region of Peninsular Malaysia, encompassing parts of Perlis and Kedah, one of the country’s major rice-producing regions [12]. In paddy cultivation, herbicides, insecticides, and fungicides were frequently used to control weeds, pests, and diseases, thereby enhancing crop yields. During the rainy season or with irrigation, excess water from paddy fields transported agricultural pollutants, including silt and pesticide residues, into surrounding water bodies and shallow groundwater aquifers [12].

In areas with poor drainage infrastructure or where agricultural practices exacerbated soil erosion, pesticide-laden runoff more easily infiltrated soil and groundwater. Moreover, pesticides had the ability to persist in groundwater, contaminating drinking water sources and negatively impacting aquatic ecosystems [13]. Urban and residential areas also contributed to pesticide contamination in groundwater [14]. Pesticides applied in residential lawns, parks, gardens, and golf courses for pest control were often washed away by excessive watering or rainfall, seeping into the soil and eventually reaching groundwater [14]. Stormwater runoff from metropolitan areas further carried pesticides into surface waters and groundwater recharge zones [14]. Additionally, large forested regions were sometimes treated with pesticides for forest management, such as herbicides for vegetation control or chemicals for pest eradication

[15]. Rainfall facilitated the washing of these chemicals from treated areas, leading to soil infiltration and runoff. Aquifer quality in forested and downstream areas was thus impacted by non-point source contamination from forestry activities. The application of pesticides along transportation corridors such as highways and railways, to control roadside weeds also likely contributed to non-point source pollution. During heavy rainfall, pesticides sprayed along these routes overflowed or leached into nearby soils and groundwater [15]. Lastly, pesticides entered groundwater through atmospheric deposition, where pesticide particles or vapors settled from the air onto land surfaces and subsequently infiltrated the soil [16]. This process affected large geographic areas and was influenced by factors such as pesticide volatility, wind patterns, and the distance from the site of application, as illustrated in Figure 1.

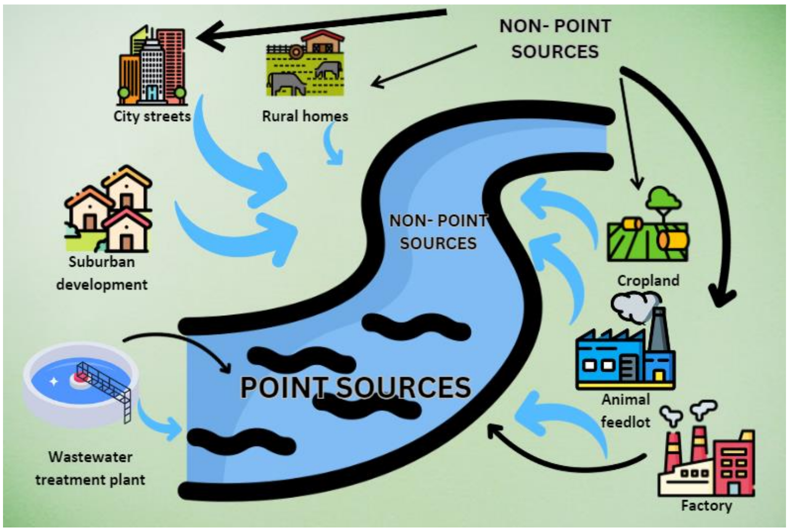

Figure 1. Point and non-point sources of pesticides.

2.3. Distribution and movement.

The interaction between pesticides and soil was the primary step in their transport into groundwater. Pesticide absorption, retention, and mobility were influenced by soil variables such as structure, pH, mineral composition, and organic matter content [17]. Pesticides could bind to soil particles or organic matter, which reduced their mobility and enhanced their persistence in the soil. These soil characteristics affected the distribution of chemicals within the soil profile, with different pesticides exhibiting varying degrees of adsorption and movement [17]. Hydrogeological factors such as aquifer properties, hydraulic connectivity, groundwater flow rates, and recharge zones, also affected the distribution of pesticides in groundwater. As water percolated through the soil profile, pesticides could migrate vertically until reaching the groundwater table [18]. The rate and direction of groundwater flow were determined by factors such as permeability, aquifer porosity, and connectivity, which, in turn, influenced pesticide dispersion in groundwater [18]. Leaching from the soil surface was the major pathway through which pesticides entered groundwater [19]. Once water infiltrated the soil profile, pesticides applied to agricultural fields or other land surfaces moved downward and reached saturated zones where groundwater was present. The distribution and movement of pesticides into groundwater are summarized in Figure 2.

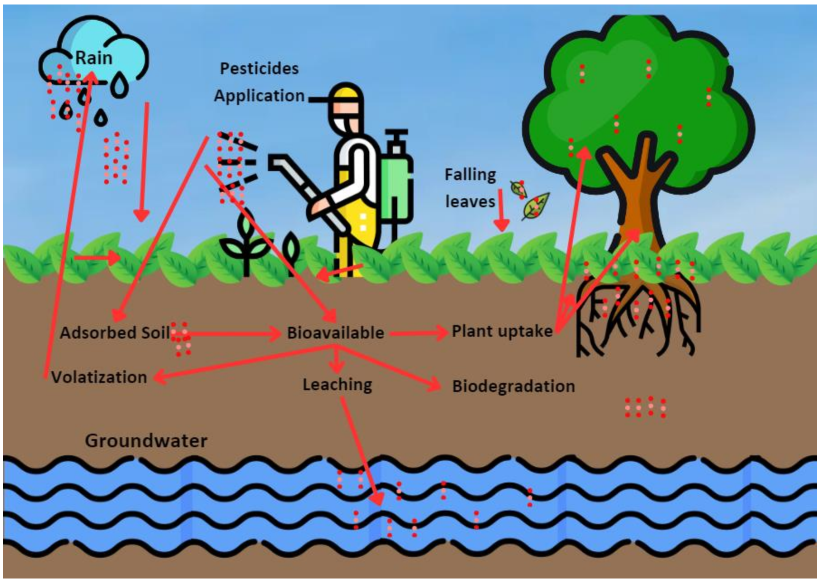

Figure 2. Distribution and movement of pesticides in groundwater.

Numerous factors, such as soil moisture content, soil structure, chemical properties, and groundwater table depth, impacted how pesticides migrated through the soil [19]. Groundwater was most likely to be polluted by pesticides with higher solubility in water and lower adsorption to soil particles. Additionally, hydraulic gradients and groundwater flow rate regulated the movement of pesticides in groundwater once they entered the saturated zones [20]. Pesticides were able to travel horizontally and vertically within aquifers by following the paths taken by groundwater. Aquifer heterogeneity, preferential flow paths, and groundwater flow directions were among the factors that affected the movement of pesticides in groundwater [20]. Long-distance movement of pesticides via groundwater systems increased the likelihood of polluting remote wells or water sources. Transportation processes along with photochemical, microbiological, and hydrolytic degradation impacted the fate of chemicals in groundwater. Pesticides, particularly those with high persistence and low susceptibility to degradation, could remain in groundwater for extended periods. Nonetheless, progressive degradation processes reduced the concentration of pesticides, thereby altering their ecological fate and behavior in groundwater systems [21].

3. Application of Mathematical Model in Groundwater System

In Malaysia, the investigation and control of pesticide contamination in groundwater systems were significantly supported by mathematical models (Table 1). By utilizing these models, researchers and decision-makers were able to predict the transport and fate of pesticides in the subsurface environment and understand the interactions among pesticides, soil, and groundwater. Pesticide movement in groundwater was simulated using mathematical models that accounted for land use patterns, hydrogeological conditions, chemical properties, and soil characteristics. The spatial and temporal distribution of pesticides in groundwater was predicted using transport models such as advection-dispersion models or groundwater flow models integrated with pesticide transport components [22]. These models helped identify pesticide-contaminated areas and assessed the effectiveness of mitigation efforts. Potential hazards related to pesticide contamination in groundwater were assessed using mathematical models. To predict the adverse effects of pesticides on the environment and human health, risk analysis models were used, incorporating data on pesticide usage, hydrogeological characteristics, environmental conditions, and exposure pathways. The use of mathematical models facilitated the prioritization of monitoring campaigns and regulatory changes to mitigate risks to human health and groundwater quality [23]. Moreover, these models aided in identifying potential sources of groundwater contamination.

To determine contamination sources, source recognition models analyzed the spatial and temporal distribution patterns of pesticides in groundwater [24]. This practice was critical to reducing future pesticide contamination and implementing effective mitigation measures. Scenario analysis using mathematical models simplified the evaluation of how different land use practices, pesticide management strategies, and climate change scenarios could influence pesticide-induced groundwater pollution [25]. Decision-makers were able to promote sustainable pesticide usage and conserve groundwater resources by exploring various strategies [26]. The development of regulations and policies aimed at reducing pesticide contamination was supported by these models, as they guided decisions regarding pesticide application, the establishment of buffer zones near water bodies, and the implementation of best management practices in agriculture [26].

Table 1. Application of mathematical models in groundwater system.

No	Model	Case study	Country	Reference
1	Flow Models	Peninsular Malaysia Groundwater Flow Model	Malaysia	[27]
2	Contaminant Transport Models	Langat River Basin Contaminant Transport Model	Malaysia	[27]
3	Integrated Surface Water-Groundwater Models	Johor Integrated Surface Water-Groundwater Model	Malaysia	[27]
4	Modular Three-Dimensional Finite-Difference Groundwater Flow Model (MODFLOW)	Chiang Mai Basin	Thailand	[28]
5	Seawater and Freshwater Transport (SEAWAT)	Upper Ring River Basin	Thailand	[28]
6	Modflow-2005 with Multi-species Transport in 3-Dimensional Model System (MODFLOW-2005 with MT3DMS)	Eastern Region	Thailand	[28]
7	Photoreactive Equilibria in Aqueous Systems (PHREEQC)	Chiang Mai Basin	Thailand	[28]
8	Numerical Groundwater Flow Models	Mekong Delta	Vietnam	[29]
9	Contaminant Transport Models	Binh Duong Province	Vietnam	[29]
10	Integrated Surface Water-Groundwater Models	Red River Delta	Vietnam	[29]
11	Data-Driven Models	Ho Chi Minh City	Vietnam	[29]
12	Numerical Groundwater Flow Models	Central Ground Water Board (CGWB)	India	[30]

4. Pesticides Evolutions to the Community

Groundwater can be considered the foundation of life, as it supplies fresh water to the entire ecosystem including a diverse array of flora and fauna, as well as humans. Groundwater contamination poses significant challenges, as both the quality and quantity of groundwater are vital at local and global scales.

4.1. Pesticide movement in environment.

Pesticides can enter the environment when they are sprayed on target plants or improperly disposed of. Once released, they undergo processes such as transport (or migration) and degradation. The breakdown of pesticides in the environment leads to the formation of new compounds. Pesticides are transported through various mechanisms, including adsorption, leaching, volatilization, spray drift, and runoff, which carry them from the target site to other environmental media or non-target organisms [31]. The different chemical classes of pesticides influence how they behave in the environment. For example, organochlorine compounds such as DDT exhibit low acute toxicity but have a strong tendency to bioaccumulate in tissues and remain harmful over long periods. Although their sale has been banned in most countries, their persistent nature means that residues continue to exist in the environment for many years. In contrast, organophosphate insecticides, despite having short half-lives, cause high acute toxicity in animals [32].

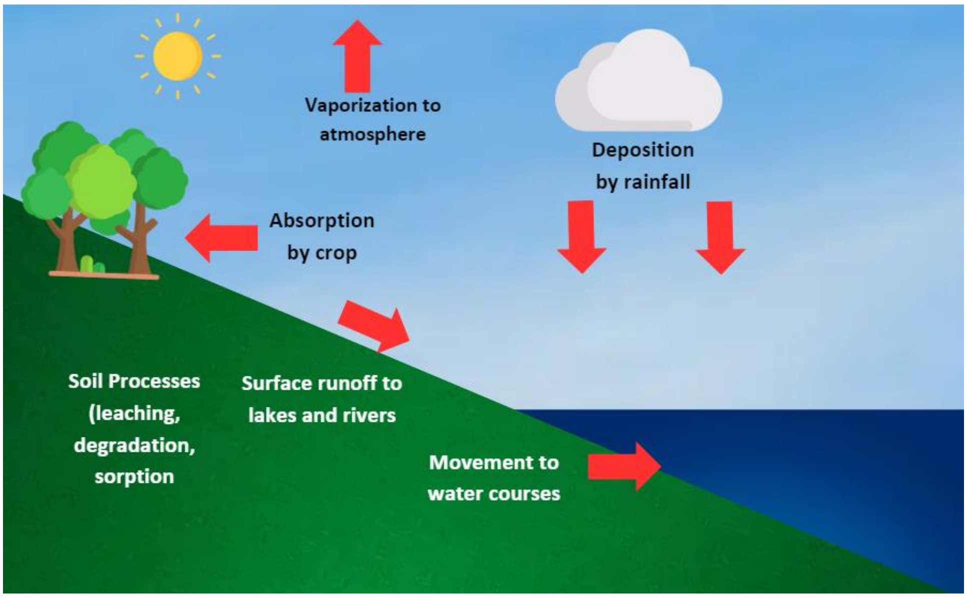

Figure 3. Pesticide behaviour in the natural environment.

4.2. Pesticides degradation.

Following their application to target organisms, pesticides are degraded by light, chemical reactions, or microbial activity. The degradation process may take hours, days, or even years, depending on environmental conditions and the chemical properties of the pesticide. These degradation mechanisms produce distinct metabolites and determine how long pesticides persist in the soil. The concept of pesticide half-life in the environment has been well documented [33]. For example, the primary metabolite of chlorpyrifos, 3,5,6-trichloro-2-pyridinol (TCP), is significantly more mobile and hazardous than chlorpyrifos itself. In many regions, both chlorpyrifos and its degradation byproducts have been commonly detected in soils, sediments, and groundwater. These substances are considered endocrine disruptors and may pose health risks to humans.

Pesticide degradation occurs through three main pathways: biological, chemical, and physical (Table 2). Microbial degradation refers to the breakdown of pesticides by microorganisms such as bacteria and fungi [34]. For instance, biodegradation is the primary mechanism for the breakdown of niclosamide in natural environments, as certain anaerobic and adapted aerobic bacteria are highly effective at degrading it. Several factors influence microbial pesticide degradation, including temperature, oxygen availability, soil moisture, pH, and soil porosity. For example, soil pH is a key factor in the enantioselective degradation of benalaxyl, with higher pH values promoting increased breakdown.

Chemical degradation, or chemical deterioration, occurs when soil-based chemical reactions transform pesticides into other compounds [35]. Ultraviolet (UV) radiation also plays a major role in breaking down pesticide molecules on soil surfaces, as it is a continuous chemical process. The rate and extent of chemical degradation are affected by factors such as temperature, pH, moisture content, and pesticide-soil binding. Photodegradation, the breakdown of pesticides by sunlight, is another important degradation pathway [36]. All insecticides are susceptible to photodegradation to varying degrees, and the rate depends on the intensity of the light source, duration of exposure, and the physicochemical properties of the pesticide [36]. For instance, when exposed to light, niclosamide may hydrolyze into 2-chloro-4-nitroaniline and 5-chlorosalicylic acid [34].

Table 2. The effectiveness of the approach according to type of pesticides.

Approach	Type of Pesticides Treated	Effectiveness	Reference
Biological			
Bioremediation	Organophosphate insecticides (e.g., Chlorpyrifos)	Highly effective, ranging from 40% to over 90%	[37]
Phytoremediation	Heavy metals and persistent organic pollutants (POPs), e.g., DDT	Effective, ranging from 20% to 80%	[37]
Chemical			
Chemical Oxidation / Reduction	Chlorinated pesticides (e.g., Lindane)	Effective, ranging from 60% to 90%	[38]
Coagulation / Flocculation	Herbicides (e.g., Atrazine); Fungicides (e.g., Chlorothalonil)	Effective, ranging from 70% to 95%	[38]
Membrane Processes (Reverse Osmosis, Ultrafiltration, Nanofiltration)	Herbicides (e.g., Atrazine); Insecticides (e.g., Chlorpyrifos)	Highly effective, exceeding 95% to 99%	[38]
Physical			
Adsorption	Herbicides (e.g., Glyphosate); Insecticides (e.g., Imidacloprid)	Highly effective, exceeding 90%	[39]
Filtration	Herbicides (e.g., Atrazine); Insecticides (e.g., Diazinon)	Effective, ranging from 60% to 90%	[39]

4.2.1. Sorption.

When herbicides were sprayed, only a very small percentage of the pesticides exhibited a protective effect against plant diseases. In contrast, a significant portion entered the soil, leading to severe contamination. Due to the attraction between chemicals and soil particles, a mechanism known as the sorption process caused pesticides to attach to soil particles [40]. Furthermore, the standard batch equilibration method was used to derive adsorption isotherms, which were then employed to evaluate pesticide retention in environmental media. This soil adsorption process was influenced by several factors. Key soil parameters that affected pesticide adsorption included pH, organic matter content, and soil amendments. Additionally, compared to coarse, sandy soils, those rich in organic matter or clay were substantially more adsorptive due to a greater number of binding sites or larger particle surface area. For instance, the clay and organic matter content of soil impacted endosulfan's capacity for adsorption and desorption [41].

In this study, the sorption, desorption, and mobility of strobilurin fungicides were examined in three Chinese soils: Jiangxi red soil, Taihu paddy soil, and Northeast China black soil. The primary factors influencing adsorption and desorption of fungicides were soil properties such as pH, cation exchange capacity (CEC), and soil organic matter (SOM) [42]. Soil moisture also influenced pesticide adsorption. As water molecules in wet soils competed with insecticides for binding sites, dry soils typically absorbed more pesticides. Another factor affecting ammonium nitrogen adsorption was temperature. The adsorption of DDT in sediments was also influenced by humic acid colloids. Certain pesticides were taken up by plants during growth due to their prolonged persistence in soil [43], which could harm crops or leave behind residues. Positively charged pesticide molecules were readily attracted to negatively charged clay particles, facilitating strong bonding [41, 43].

4.2.2 Leaching.

Pesticides were registered and used in large quantities worldwide, with some leaching into groundwater and contaminating water supplies. Numerous factors affected leaching. Soil permeability was a key factor; the more permeable the soil, the higher the likelihood of pesticide leaching. Leaching was also influenced by the adsorption coefficient (K_{oc}) and the pesticide's half-life in aerobic soil (DT50) [44]. The persistence of pesticides further affected leaching potential; pesticides with low persistence remained in soil briefly and were less likely to leach. For instance, imidacloprid had high environmental persistence (DT50 in soil = 187 days) [45].

Meteorological factors such as average annual temperature and rainfall significantly influenced leaching. Precipitation played a major role in downward water movement, facilitating pesticide transport into groundwater. Temperature affected evapotranspiration, which in turn influenced leaching dynamics. Soil features—texture and organic content—were also important, with texture being the most significant factor governing water movement and pesticide transfer. Additionally, organic matter content, pH levels, and anaerobic microbial populations influenced the degradation of compounds like phenazines. Research showed that soil pH had a strong negative correlation with pesticide adsorption, while weaker correlations existed with the carbon and nitrogen content in SOM and the organo-mineral complex (TOC, clay, surface area). Although particle size showed no direct correlation, factor analysis revealed a multidimensional link between these variables and CEC [47]. The impact of crop residues on soil water and temperature regimes also had to be considered, particularly in relation to evaporation rates.

4.2.3. Spray drift.

Spray drift referred to the aerial movement of pesticide droplets from the application site, contaminating food and the environment. For example, leaching, spray drift, and runoff exposed aquatic ecosystems to residues like chlorpyrifos (ChF), which were toxic to aquatic life. Researchers studied the effects of ChF exposure on oxidative stress enzymes and histological changes in vital organs of tilapia, finding that even sub-lethal concentrations caused damage [48].

Another example was the use of low-volume, fine and very-fine-droplet applications via unmanned aerial vehicles (UAVs) in commercial agriculture. These allowed for efficient pesticide delivery and water conservation. However, spray drift from UAVs, especially fine droplets produced by spinning disc nozzles, remained under-examined, raising regulatory and environmental concerns. A study tested the drift potential of three volume median diameters

(VMDs: 100, 150, and 200 µm) using a commercial quadcopter with centrifugal nozzles at varying wind speeds. Results showed that flight speed and altitude significantly influenced airflow dispersion patterns [49].

4.2.4. Volatilization.

Volatilization is the process by which a solid or liquid transforms into a gas. Once volatilized, pesticides could be transported from the treated surface by air currents. The extent of volatilization depended on factors such as moisture, organic matter content, soil texture, temperature, humidity, and air movement. Pesticides with higher vapor pressure were more volatile. High temperatures, low humidity, and increased air movement promoted volatilization. For instance, tropical regions had higher atmospheric dispersion of organochlorine pesticides (OCPs) compared to temperate zones [50]. Tightly adsorbed pesticides were less likely to volatilize. Research showed that surface waters could become a major source of OCP exposure to humans through volatilization. One study monitored chlorpyrifos and its breakdown product, chlorpyrifos oxon, in soil, air, and plant tissue for 21 days after application to a purple tansy field. The estimated chlorpyrifos concentrations were five times higher than expected, highlighting the need for improved methods to estimate pesticide volatilization emissions from agricultural lands [51].

4.2.5. Surface runoff.

Surface runoff is the process by which pesticides travel via water flow over a sloped surface. Pesticides can be carried in dissolved form or attached to eroded soil particles. Factors influencing runoff included slope, soil structure, moisture content, rainfall intensity and timing, and irrigation practices. Runoff occurred when water was applied faster than the soil could absorb it. Over-irrigation led to excess surface water and thus greater pesticide runoff. The discharge of pesticides via runoff polluted streams, ponds, lakes, and wells, with harmful effects on humans, animals, and plants [52].

4.3. Impact to living organisms.

4.3.1. Uptake, bioaccumulation and biotransformation.

There are several methods for pesticides to enter living things, including by ingestion, breathing, and skin absorption. In addition to directly consuming food, water, or prey contaminated with pesticides, organisms can also absorb pesticides through their skin or respiratory surfaces. Pesticides can travel throughout the body through the circulatory system or other transport systems once they are inside the organism [53].

Bioaccumulation is the term for the gradual build-up of pesticides in the tissues of creatures that they come into contact with. This happens when the rate of pesticide absorption by an organism surpasses its rate of metabolism or excretion. Pesticides that are lipophilic, or fat-soluble, tend to accumulate in fatty tissues, whereas pesticides that are water-soluble might accumulate in organs like the kidneys or liver. Higher trophic levels exhibit more intense bioaccumulation since pesticides bio-magnify as they ascend the food chain [54].

Pesticides can pass through metabolic processes while they are within an organism. During these processes, enzymes catalyse chemical reactions that turn the pesticide molecules into metabolites. Pesticides can be activated or detoxified by metabolism, which changes the toxicity and duration of the chemicals inside the body. Depending on the chemical structure of pesticide and the metabolism capacity of an organism, several biotransformation processes apply [55].

4.3.2. Toxicity and adverse effects

Through a variety of processes, such as the disruption of biochemical pathways, interference with cellular functioning, and damage to tissues and organs, pesticides can have harmful effects on living things (Table 3). Shortly after being exposed to high pesticide concentrations, an individual may experience acute poisoning, which can include symptoms including nausea, vomiting, headaches, disorientation, respiratory distress, convulsions, and in extreme circumstances, death. Long-term health impacts, such as neurological diseases, reproductive abnormalities, endocrine disruption, immune system suppression, respiratory illnesses, and some forms of cancer, may arise from chronic exposure to low concentrations of pesticides [56].

Table 3. Health impacts according to type of pesticides in different countries.

No	Pesticide	Health impact	Country	References
1	Organophosphate Insecticides (examples: Chlorpyrifos, malathion, diazinon)	Causes nausea, dizziness, headaches, respiratory problems and in severe cases, paralysis or death	Malaysia	[57]
2	Pyrethroid Insecticides (examples: Permethrin, cypermethrin, deltamethrin)	Causes irritation to the skin, eyes and respiratory system. High-level exposure lead to itching, burning sensations, dizziness, nausea and respiratory distress	Malaysia	[57]
3	Organochlorine Pesticides (examples: DDT, Endosulfan, Lindane)	Causes cancer, reproductive disorders, neurotoxicity and disruption of endocrine function	Malaysia	[58]
4	Herbicides (examples: Glyphosate, Paraquat, Atrazine)	Lead to skin and eye irritation, respiratory problems, gastrointestinal issues and in severe cases, organ damage	Malaysia	[57]
5	Fungicides (examples: Mancozeb, Chlorothalonil, Carbendazim)	Causes skin and eye irritation, respiratory problems and gastrointestinal issues	Malaysia	[57]
6	Neonicotinoid Insecticides (examples: Imidacloprid, Clothianidin)	Causes neurological effects and developmental toxicity	United States	[58]
7	Paraquat Herbicide	Leads to respiratory problems, kidney damage, Parkinson’s disease and increased risk of certain cancers	Brazil	[58]
8	Organochlorine Pesticides (examples: DDT, Lindane)	Causes neurological disorders, reproductive problems, immune system suppression and carcinogenic effects	China	[58]
9	Carbamate Insecticides (examples: Carbofuran, Methomyl)	Causes nausea, vomiting, dizziness, headaches and in severe cases, respiratory failure or death	Kenya	[57]
10	Organophosphate and Carbamate Insecticides	Leads to cholinergic toxicity, respiratory depression and neurological effects. Chronic exposure causes neurodevelopmental disorders and cancer	Thailand	[58]

Low-dose pesticide exposure over an extended period of time can have long-term negative consequences on health. Chronic pesticide exposure has been linked to a number of health issues, including immune system suppression, endocrine disruption, respiratory conditions, neurological disorders example Parkinson's disease and cognitive impairments, reproductive disorders such as infertility, miscarriage, and birth defects, and some cancers like leukaemia, lymphoma, and cancers of the breast, prostate, and lung [57].

Early childhood or crucial foetal development is a time when pesticide exposure can disrupt normal growth and development, resulting in behavioural problems, developmental abnormalities, and cognitive impairments. Additionally, pesticides have the potential to interfere with fertility in a woman, hormone levels, and pregnancy outcomes in both men and females. Furthermore, there is evidence linking prenatal pesticide exposure to a higher chance of unfavourable birth outcomes, including low birth weight, preterm birth, and congenital abnormalities [58].

5. Remediation Technologies for Pesticides Movement in Groundwater

5.1. Pump-and-treat systems.

One remediation technique that was frequently used to address pesticide contamination in groundwater was the pump-and-treat method (Table 4). In this method, contaminated groundwater was extracted from wells using pumps, treated to remove or degrade pesticides, and then either reinjected into the aquifer or discharged into surface water bodies. Before installing a pump-and-treat system, a comprehensive site characterization was conducted to assess the extent and distribution of pesticide contamination in the groundwater. Groundwater monitoring wells were strategically placed to collect data on groundwater flow rates, pollutant concentrations, hydrogeological characteristics, and other relevant factors [59].

Pump wells were positioned at key locations within the contaminated groundwater plume to extract water for treatment. The placement and number of these wells depended on variables such as the extent of contamination, direction of groundwater flow, and aquifer permeability. Using submersible pumps or other pumping equipment, contaminated groundwater was transported from the extraction wells to surface treatment facilities. At these facilities, the groundwater underwent a series of treatment processes designed to remove or degrade pesticides. Treatment technologies included filtration using granular activated carbon (GAC) filters, which adsorbed pesticide molecules [60]. Additionally, chemical oxidants such as hydrogen peroxide were employed to chemically degrade pesticides.

Biodegradation by microorganisms through natural metabolic processes also contributed to the breakdown of certain pesticides [60]. After treatment, the purified groundwater was either released into surface water bodies, if permitted by regulations, or reinjected into the aquifer via reinjection wells. Reinjection helped restore groundwater levels and hydraulic gradients, thereby promoting natural attenuation processes and dilution of residual contaminants.

To ensure continued effectiveness, pump-and-treat systems required regular operation and maintenance. This included managing flow rates, evaluating treatment performance, monitoring groundwater levels, and maintaining pumping equipment, monitoring wells, and treatment infrastructure [61]. Long-term monitoring of pollutant concentrations and groundwater quality was essential for assessing the effectiveness of the system. Performance evaluations helped determine whether remediation goals were being achieved and whether system modifications were necessary. However, installation and maintenance of pump-and-treat systems could be costly, especially for long-term projects involving extensive plumes or complex hydrogeological conditions [62].

Table 4. Comparison of remediation technologies for pesticide movement in groundwater.

Technology	Description	Advantages	Disadvantages	References
Pump-and-Treat	Extraction of contaminated groundwater, surface treatment, then reinjection/discharge.	- Effective for diverse contaminants - Well-established and proven - Adjustable treatment parameters	- High energy and maintenance cost - Long remediation time - Less effective for DNAPLs	[59–63]
In Situ Chemical Oxidation (ISCO)	Injection of oxidants (e.g., H_2O_2, $KMnO_4$) into the subsurface to chemically degrade contaminants.	- Rapid degradation of pollutants - Effective even in low-permeability zones - Reduces contaminant mass and plume spread	- Risk of harmful byproducts - Limited contact in heterogeneous subsurface - Requires precise site characterization	[64–68]
In Situ Biological Remediation	Uses microbes to degrade pollutants via aerobic/anaerobic metabolism, co-metabolism, or co-oxidation.	- Environmentally friendly and low-energy - Cost-effective - Minimal site disturbance	- Slower process - Less effective for highly toxic/mobilized contaminants - Requires close monitoring	[69–73]
Phytoremediation	Uses plants to absorb, accumulate, or degrade pollutants in soil and groundwater.	- Sustainable and aesthetic - Low-cost for large areas - Enhances ecological value	- Slow remediation process - Limited to certain contaminants - Risk of contaminant transfer through food chain	[74–77]

Pump-and-treat systems presented both advantages and disadvantages. One major advantage was their applicability to a wide range of contaminants, including metals, nutrients, and organic compounds, making them suitable for various contamination scenarios. Moreover, their long history of use and research had resulted in well-established designs and principles, making them a reliable choice for groundwater remediation projects [63]. These systems also offered precise control over treatment parameters such as flow rates, residence time, and treatment media within a controlled setting.

On the other hand, pump-and-treat systems required continuous energy input for pumping, treatment, and monitoring, resulting in high operational and maintenance costs. Over time, as treatment progressed and groundwater flow decreased, these costs could increase. In cases involving large or complex contamination plumes, remediation with pump-and-treat systems often became a lengthy process, potentially taking years or decades to achieve cleanup objectives [62]. Furthermore, the treatment of dense non-aqueous phase liquids (DNAPLs), such as chlorinated solvents that persisted below the surface as immiscible pollutants, was often ineffective using this method [63].

5.2. In situ chemical oxidation (ISCO).

The second remediation technique is by in situ chemical oxidation (ISCO). By injecting chemical oxidants directly into the subsurface to break down or change organic pollutants into less harmful forms, ISCO is a remediation technology used to clean polluted soil and groundwater. Organic pollutants found in soil or groundwater are reacted with by chemical oxidants that are injected into the polluted subterranean environment in ISCO. Target pollutants undergo oxidation processes with the help of the chemical oxidants, which produce very reactive oxidising species which is hydroxyl radicals. After that, the organic pollutants are reduced to simpler, less hazardous substances for example, carbon dioxide, water, and inorganic salts by oxidation processes [64]. There are some chemical oxidants that can be used in ISCO applications such as ozone (O_3), hydrogen peroxide (H_2O_2), potassium permanganate ($KMnO_4$) and sodium persulfate ($Na_2S_2O_8$). This oxidant is influenced by several elements, including the kind of pollutants, the site's features, and the treatment goals. Each oxidant has unique qualities and reactivity characteristics. Usually, infiltration galleries, direct-push technologies, or injection wells are used to introduce chemical oxidants into the subsurface. To bring the oxidant solution to the target region, injection wells are positioned strategically throughout the polluted zone. Along with using specialized equipment, direct-push modern technology injects the oxidant solution into the subsurface, making it possible for exact oxidant circulation in the contaminated area [65].

The chemical oxidant injection ensures continuos contacts with the contaminants by dispersing and mixing with the soil or groundwater after the injection. Aquifer features, groundwater flow rate and injection strategy are several of the factros that influence the oxidant flow and mixing. The chemical oxidants trigger oxidation responses when they enter contact with the organic contaminant, which causes the contaminant to weaken or alter. Direct oxidation by the oxidant varieties as well as the manufacturing of responsive intermediates like hydroxyl radicals are two ways in which oxidation may take place. After that, oxidation reactions include the breakdown of complex organic molecules into simpler and less hazardous chemicals by means of chemical bond breaking, dehydrogenation, and hydroxylation [66]. Groundwater quality, pollutant concentrations, and oxidant dispersion must be monitored throughout ISCO treatment in order to evaluate treatment efficacy. In order to monitor changes in pollutant concentrations over time and gauge the degree of contaminant degradation, performance evaluation entail the collection and analysis of soil and groundwater samples. For ISCO applications to produce the intended treatment results, site characterization, and treatment parameter optimisation are necessary. However, there will be challenges such as oxidant delivery, the potential for secondary reactions and mobilization of metals and other contaminants [65].

In situ chemical oxidation (ISCO) have its advantage and disadvantages. One of the advantages is that compared to pump-and-treat systems or natural attenuation, in situ chemical oxidation can quickly degrade contaminants in soil or groundwater, expediting remediation. Furthermore, chemical oxidants have the potential to improve mass transfer procedures even in limited permeability zones and DNAPL source locations by making reactive species more readily available and encouraging oxidation interactions with target pollutants. By reducing the quantity of contaminants below the surface, in situ chemical oxidation can attenuate groundwater plumes over time and lessen their source [67]. However, if in situ chemical oxidation is not adequately managed or regulated, it can result in by products or intermediates that are hazardous, persistent, or mobile and might endanger human health or the environment. Moreover, the regions where enough contact between the chemical oxidant and pollutants can be obtained, especially in poor permeability or heterogeneous subsurface conditions, can be the limit of the efficiency of in situ chemical oxidation. Accurate site characterization is necessary for the effective use of in situ chemical oxidation in order to determine the hydrogeological characteristics, redox conditions, and distribution of contaminants. In complicated subsurface settings, this process might be difficult or unpredictable [68].

5.3. In situ biological remediation.

The third remediation technique is by in situ biological remediation which is also known as bioremediation. Bioremediation is a sustainable and economical method of reducing soil and groundwater contamination by utilising the ability of microorganisms to break down or reduce pollutants into less harmful forms. Bioremediation breaks down natural pollutants in soil or groundwater by using naturally-occuring microorganisms such as fungi and bacteria. Organic contaminants are broken down by microorganisms into less complex and safer forms via biochemical procedures which they utilize as a source of energy and nourishment. Many metabolic procedures consisting of co-metabolism, co-oxidation and both anaerobic and aerobic degradations, are part of the biodegradation of organic contaminants [69]. When there are oxygen, existing organic contaminants undertake aerobic biodegradation which breaks them down into CO_2, water and biomass. On the other hand, methane hydrogen sulphide, and other reduced substances are usually generated throughout anaerobic biodegradation, which occurs when there is no oxygen and consists of the microbial reduction of contaminants. Enzymes generated by microorganisms during the metabolism of other chemicals are responsible for the breakdown of pollutants through co-metabolism and co-oxidation [70].

There are some technologies that can be achieved when using bioremediation such as bio-stimulation and bioaugmentation. Both aims to encourage the natural activities of microorganisms in the polluted environment to accelerate the breakdown of pollutants. Through microbial metabolism, bioremediation can efficiently breakdown a variety of pesticides in the setting of pesticide pollution, including fungicides, herbicides, and insecticides. Pesticides can be broken down into simpler, non-toxic chemicals by microorganisms that can break them down into carbon and energy sources. Ex situ which is in above-ground treatment systems or in situ which is in the soil or groundwater bioremediation can be used to remediate pesticide-contaminated areas [71]. In situ biological remediation can be implemented successfully if site-specific factors such as hydrogeology, pollutant properties and microbial ecology are carefully taken into account. Furthermore, enhancing biodegradation rates and achieving the intended treatment outcomes require optimisation of treatment parameters such nutrient supplement, oxygenation, pH correction, and microbial inoculation [72]. However, it is crucial to monitor microbial activity, pollutant concentrations, and groundwater quality in order to evaluate the efficacy of treatment. Sampling and analysing soil and groundwater samples can be part of performance evaluation in order to monitor variations in contaminant concentrations over time.

In situ biological remediation have its own advantages and disadvantages. One of its advantages is that since in situ biological remediation uses natural processes and microorganisms to break down contaminants, it requires less energy and chemical additions from outside sources, making it a sustainable method [72]. Moreover, bioremediation is a cost-effective alternative for places with limited funds or resources since it has lower start-up and operating expenses than standard clean-up techniques. Other than that, since in situ biological remediation entails injecting nutrients, oxygen, or microbial inoculants straight into the subsurface without the need for significant excavation or disturbance, it can be carried out with little disruption to the site's natural circumstances. However, compared to other remediation techniques, biological remediation procedures can be comparatively slow which take weeks, months, or even years to accomplish the intended cleaning outcomes especially for complicated or highly polluted areas [73]. Although many organic pollutants can be successfully degraded by in situ biological remediation, it might not be as effective for locations with contaminants that are highly mobile and harmful. Lastly, when microbial activity only partially destroyed contaminants, biological remediation produce rebound effects, which calls for close monitoring and control.

5.4. Phytoremediation.

Phytoremediation is a specific type of bioremediation that uses plants to eliminate and break down pollutants in soil, water, or air. It is based on certain capacity of plants to absorb, store, and transform contaminants inside their tissues. These plants are referred to as hyperaccumulators or phytoremediators. There are different types of phytoremediation such as phytoextraction, phytodegradation and others. Plants are used in all forms of phytoremediation to help remove pollutants from the air, water, or soil. Through a variety of physiological and biochemical processes, plants are essential for the absorption, accumulation, transformation, and degradation of pollutants [74]. The ideal plant types should be picked for phytoremediation for it to be effective. These varieties have the required remediation features such as deep-rooted, rapid growth and also high biomass outcome. Plants referred to as hyperaccumulators work for phytoextraction since they keep high concentrations of pollutants in their cells without showing noticeable phytotoxic results [75].

Factors consisting of soil composition, environment, contaminant concetration, and treatment objectives need to be determined when selecting plants. Depending on the conditions of the contaminated areas and the goals of the remediation process, either on-site or off-site phytoremediation can be used [75]. Planting proper plants straight in infected areas permits plants to gradually uptake and eliminate toxic substances through a natural process referred to as in-position phytoremediation. Whereas expanding plants in synthetic marshes, nurseries, or various other controlled conditions where contaminated soil or water presented for treatment is called ex-situ phytoremediation. Furthermore, it is crucial to monitor soil and water quality, plant health, and pollutant absorption in order to evaluate the efficacy of phytoremediation. In order to monitor changes in pollutant concentrations over time and gauge the extent of contaminant removal or degradation, performance evaluation involves routine sampling and analysis of plant tissues, soil, and groundwater.

Phytoremediation have its own advantages and disadvantages. One of the advantages is that the natural and sustainable method of phytoremediation minimises the need for intrusive remediation methods or disruptive excavation by using plants and the bacteria they are linked with to clean contaminated areas. Given that it frequently requires smaller initial capital investment and ongoing operating costs, phytoremediation is more affordable than standard clean-up techniques, especially for large-scale and long-term projects. Moreover, through plant establishment, habitat restoration, and the provision of ecosystem services like carbon sequestration, erosion control, and wildlife habitat, phytoremediation can improve the aesthetic appeal and ecological function of contaminated areas [76]. However, when it comes to achieving considerable reductions in pollutant concentrations especially for persistent or resistant contaminants, phytoremediation procedures can be comparatively slow which is taking months or even years. Furthermore, for contaminants with high toxicity, mobility, or bioavailability, phytoremediation cannot be as successful or practical depending on the pollutants, the location, or the required regulations. Certain types of phytoremediation cause pollutants to build up in plant tissues, which might increase the risk of biomagnification and the spread of contaminants up the food chain, especially for persistent organic pollutants and heavy metals [77].

6. Conclusions

The presence of pesticides in Malaysia's groundwater highlights the urgent need for comprehensive and immediate action to address this critical environmental and public health issue. The unregulated and widespread use of pesticides in agriculture, combined with poor management practices, has significantly contributed to groundwater contamination across the country. This pollution poses a serious threat to both environmental ecosystems and human health due to the potential risks associated with exposure to pesticide-contaminated water. To reduce pesticide contamination in groundwater, coordinated efforts among all stakeholders are essential. These efforts should include the implementation of effective control measures, such as establishing nationwide monitoring programs and enforcing stringent pest management regulations. Beyond regulatory actions, emphasis must be placed on promoting sustainable agricultural practices. Approaches like organic farming and integrated pest management offer viable alternatives that not only reduce chemical usage but also ensure agricultural productivity remains unaffected. Furthermore, advancing research into the sources, behavior, and impacts of pesticides in groundwater systems is critical for informing evidence-based decision-making and for developing targeted mitigation strategies. Such research will enable the prioritization of interventions and support the formulation of effective, science-driven policies. In summary, collaboration among governmental bodies, agricultural stakeholders, scientists, and the public is vital to addressing pesticide pollution in Malaysian groundwater. By enacting policies that promote responsible pesticide use and by investing in research and education, Malaysia can safeguard its groundwater resources and enhance the health, well-being, and environmental sustainability for current and future generations.

Acknowledgments

The authors gratefully acknowledge the support provided by Curtin University Malaysia, the School of Engineering and Digital Science in Kazakhstan, Lhokseumawe State Polytechnic and Bina Nusantara University in Indonesia, the University of Yamanashi in Japan, the University of Birmingham in the United Kingdom, Universitas Padjadjaran in Indonesia, Pohang University of Science and Technology in the Republic of Korea, and Morgan State University in the USA for their invaluable contributions to the facilitation of this research project.

Author Contribution

Audrey Primus and Nikita Emalya contributed to the drafting of the manuscript, writing and revision, conceptualization, and data analysis. Cut Yusnar, Yureana Wijayanti, Rubiyatno, and Rega Permana were responsible for the methodology and interpretation of results. Sang Hyeok Park, Ocean Thakali, Corry Aina, Ni Putu Sri Wahyuningsih, and Nii Amarquaye Commey contributed to the research design, data collection, interpretation, as well as writing and revision of the manuscript.

Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this article.

References

[1] Andrew, J.; Ismail, N.W.; Djama, M. (2017). An overview of genetically modified crop governance, issues and challenges in Malaysia. *Journal of the Science of Food and Agriculture*, *98*(1), 12–17. https://doi.org/10.1002/jsfa.8666.

[2] Devadason, E.S.; Chandran, V.; Kalirajan, K. (2018). Harmonization of food trade standards and regulations in ASEAN: the case of Malaysia's food imports. *Agricultural Economics*, *49*(1), 97–109. https://doi.org/10.1111/agec.12398.

[3] Sass, J.B.; Colangelo, A. (2006). European Union bans atrazine, while the United States negotiates continued use. *International Journal of Occupational and Environmental Health*, *12*(3), 260–267. https://doi.org/10.1179/oeh.2006.12.3.260.

[4] Mogaji, K.A.; Lim, H.S. (2018). Development of groundwater favourability map using GIS-based driven data mining models: an approach for effective groundwater resource management. *Geocarto International*, *33*(4), 397–422. https://doi.org/10.1080/10106049.2016.1273400.

[5] Loague, K.; Corwin, D.L. (2005). Point and non point source pollution. *Encyclopedia of Hydrological Sciences*. https://doi.org/10.1002/0470848944.hsa097.

[6] Barrow, C.J.; Chan, N.W.; Masron, T.B. (2010). Farming and other stakeholders in a tropical highland: Towards less environmentally damaging and more sustainable practices. *Journal of Sustainable Agriculture*, *34*(4), 365–388. https://doi.org/10.1080/10440041003680205.

[7] Hernández, F.; Marín, J.M.; Pozo, Ó.J.; Sancho, J.V.; López, F.J.; Morell, I. (2008). Pesticide residues and transformation products in groundwater from a Spanish agricultural region on the Mediterranean Coast. *International Journal of Environmental Analytical Chemistry*, *88*(6), 409–424. https://doi.org/10.1080/03067310701724772.

[8] Bhat, V.N. (2005). Polluting facilities and environmental justice – a study. *International Journal of Environmental Studies*, *62*(1), 5–13. https://doi.org/10.1080/002072302900115 08a.

[9] Esen, F. (2013). Development of a passive sampling device using Polyurethane Foam (PUF) to measure Polychlorinated Biphenyls (PCBs) and Organochlorine Pesticides (OCPs) near landfills. *Environmental Forensics*, *14*(1), 1–8. https://doi.org/10.1080/15275922.2012.729008.

[10] Matsuo, Y.; Nakata, H.; Agusa, T.; Miyawaki, T.; Kadokami, K.; Sato, K.; Matsumoto, M.; Higuchi, T.; Nishimuta, K.; Ryuda, N.; Miyamoto, H.; Haraguchi, T.; Ueno, D. (2020). Comprehensive target analysis of micropollutants in soil at debris storage sites of the Kumamoto earthquake. *Soil and Sediment Contamination: An International Journal*, *29*(4), 452–463. https://doi.org/10.1080/15320383.2020.1738336.

[11] Blankenberg, A.-G.B.; Braskerud, B.; Haarstad, K. (2006). Pesticide retention in two small constructed wetlands: treating non-point source pollution from agriculture runoff. *International*

Journal of Environmental Analytical Chemistry, *86*(3–4), 225–231. https://doi.org/10.1080/03067310500247470.

[12] Raffar, N.; Zulkafli, Z.; Yiwen, M.; Muharam, F.M.; Rehan, B.M.; Nurulhuda, K. (2022). Watershed-scale modelling of the irrigated rice farming system at Muda, Malaysia, using the Soil Water Assessment Tool. *Hydrological Sciences Journal*, *67*(3), 462–476. https://doi.org/10.1080/02626667.2021.2022682.

[13] Zou, N.; Gu, K.; Liu, S.; Hou, Y.; Zhang, J.; Xu, X.; Li, X.; Pan, C. (2016). Rapid analysis of pesticide residues in drinking water samples by dispersive solid-phase extraction based on multiwalled carbon nanotubes and pulse glow discharge ion source ion mobility spectrometry. *Journal of Separation Science*, *39*(6), 1202–1212. https://doi.org/10.1002/jssc.201501258.

[14] Jiang, W.; Luo, Y.; Conkle, J.L.; Li, J.; Gan, J. (2015). Pesticides on residential outdoor surfaces: environmental impacts and aquatic toxicity. *Pest Management Science*, *72*(7), 1411–1420. https://doi.org/10.1002/ps.4168.

[15] Ndlovu, N.N.; Little, K.; Baillie, B.; Rolando, C. (2022). An evaluation of the environmental behaviour, fate and risk of key pesticides used in South African forest plantations. *Southern Forests: a Journal of Forest Science*, *84*(1), 83–92. https://doi.org/10.2989/20702620.2022.2045879.

[16] Li, Z.; Lin, T.; Li, Y.; Jiang, Y.; Guo, Z. (2017). Atmospheric deposition and air-sea gas exchange fluxes of DDT and HCH in the Yangtze River Estuary, East China Sea. *Journal of Geophysical Research: Atmospheres*, *122*(14), 7664–7677. https://doi.org/10.1002/2016jd026330.

[17] Gao, Y.; Zheng, H.; Xia, Y.; Chen, M.; Meng, X.; Cai, M. (2019). Spatial distributions and seasonal changes of current-use pesticides from the North Pacific to the Arctic Oceans. *Journal of Geophysical Research: Atmospheres*, *124*(16), 9716–9729. https://doi.org/10.1029/2018jd030186.

[18] Briceño, G.; Palma, G.; Durán, N. (2007). Influence of organic amendment on the biodegradation and movement of pesticides. *Critical Reviews in Environmental Science and Technology*, *37*(3), 233–271. https://doi.org/10.1080/10643380600987406.

[19] Pérez-Lucas, G.; el Aatik, A.; Vela, N.; Fenoll, J.; Navarro, S. (2020). Exogenous organic matter as strategy to reduce pesticide leaching through the soil. *Journal of Environmental Quality*, *67*(7), 934–945. https://doi.org/10.1080/03650340.2020.1768531.

[20] Perera-Rios, J.; Ruiz-Suarez, E.; Bastidas-Bastidas, P. de J.; May-Euán, F.; Uicab-Pool, G.; Leyva-Morales, J. B.; Reyes-Novelo, E.; Pérez-Herrera, N. (2022). Agricultural pesticide residues in water from a karstic aquifer in Yucatan, Mexico, pose a risk to children's health. *International Journal of Environmental Health Research*, *32*(10), 2218–2232. https://doi.org/10.1080/09603123.2021.1950652.

[21] Zhong, G.; Tang, J.; Xie, Z.; Möller, A.; Zhao, Z.; Sturm, R.; Chen, Y.; Tian, C.; Pan, X.; Qin, W.; Zhang, G.; Ebinghaus, R. (2014). Selected current-use and historic-use pesticides in air and seawater of the Bohai and Yellow Seas, China. *Journal of Geophysical Research: Atmospheres*, *119*(2), 1073–1086. https://doi.org/10.1002/2013JD020951.

[22] Gevaert, V.; Van Griensven, A.; Holvoet, K.; Seuntjens, P.; Vanrolleghem, P.A. (2008). SWAT developments and recommendations for modelling agricultural pesticide mitigation measures in river basins. *Hydrological Sciences Journal*, *53*(5), 1075–1089. https://doi.org/10.1623/hysj.53.5.1075.

[23] Eleftheriadou, D.; Luette, S.; Kneuer, C. (2019). In silico prediction of dermal absorption of pesticides – an evaluation of selected models against results from in vitro testing. *SAR and QSAR in Environmental Research*, *30*(8), 561–585. https://doi.org/10.1080/1062936x.2019.1644533.

[24] Cardenas, S.; Márquez, A.; Guevara, E. (2021). Diffusion–advection process modeling of organochlorine pesticides in rivers. *Journal of Applied Water Engineering and Research*, *11*(1), 1–22. https://doi.org/10.1080/23249676.2021.1982029.

[25] Khosravi, F.; Jha-Thakur, U. (2018). Managing uncertainties through scenario analysis in strategic environmental assessment. *Journal of Environmental Planning and Management*, *61*(5), 1–22. https://doi.org/10.1080/09640568.2018.1456913.

[26] Frelih-Larsen, A.; Chivers, C.; Herb, I.; Mills, J.; Reed, M. (2023). The role of public consultations in decision-making on future agricultural pesticide use: insights from European Union’s Farm to Fork Strategy public consultation. *Journal of Environmental Policy & Planning*, *25*(4), 476–492. https://doi.org/10.1080/1523908x.2023.2212369.

[27] Alslaibi, T.M.; Abunada, Z.; Abu Amr, S.S.; Abustan, I. (2017). Risk assessment of nitrate transport through subsurface layers and groundwater using experimental and modeling approach. *Environmental Technology*, *39*(21), 2691–2702. https://doi.org/10.1080/09593330.2017.1365936.

[28] Shrestha, S.; Shrestha, S. (2014). Evaluation of the PESTFADE model using field-measured data from a sprinkler-irrigated soybean field in Pathumthani, Thailand. *Journal of Applied Water Engineering and Research*, *2*(1), 57–69. https://doi.org/10.1080/23249676.2014.932719.

[29] Thu, T.; Everaarts, A.P.; Neeteson, J.J.; Struik, P.C. (2013). Vegetable production in the Red River Delta of Vietnam. II. Profitability, labour requirement and pesticide use. *NJAS Wageningen Journal of Life Sciences*, *67*(1), 37–46. https://doi.org/10.1016/j.njas.2013.09.003.

[30] Tomer, S. K.; Sekhar, M.; Balakrishnan, K.; Malghan, D.; Thiyaku, S.; Gautam, M.; Mehta, V. K. (2021). A model-based estimate of the groundwater budget and associated uncertainties in Bengaluru, India. *Urban Water Journal*, *18*(1), 1–11. https://doi.org/10.1080/1573062X.2020.1836237.

[31] Zhao, Y.Q.; Singleton, P.; Meredith, S.; Rennick, G.W. (2013). Current status of pesticides application and their residue in the water environment in Ireland. *International Journal of Environmental Studies*, *70*(1), 59–72. https://doi.org/10.1080/00207233.2012.752557.

[32] Gao, H.; Ma, J.; Cao, Z.; Dove, A.; Zhang, L. (2010). Trend and climate signals in seasonal air concentration of organochlorine pesticides over the Great Lakes. *Journal of Geophysical Research*, *115*(D15). https://doi.org/10.1029/2009jd013627.

[33] Conde-Avila, V.; Ortega-Martínez, L. D.; Loera, O.; El Kassis, E. G.; Dávila, J. G.; Valenzuela, C. M.; Armendáriz, B. P. (2021). Pesticides degradation by immobilised microorganisms. *International Journal of Environmental Analytical Chemistry*, *101*(15), 2975–3005. https://doi.org/10.1080/03067319.2020.1715375.

[34] Casas López, J.L.; Cabrera-Reina, A.; Gómez, E.; Ballesteros Martín, M.M.; Malato, S.; Sánchez Pérez, J.A. (2010). Integration of solar photocatalysis and membrane bioreactor for pesticides degradation. *Separation Science and Technology*, *45*(11), 1571–1578. https://doi.org/10.1080/01496395.2010.487465.

[35] Charalampous, A.C.; Miliadis, G.E.; Koupparis, M.A. (2015). A new multiresidue method for the determination of multiclass pesticides, degradation products and PCBs in water using LC–MS/MS and GC–MS(n) systems. *International Journal of Environmental Analytical Chemistry*, *95*(13), 1283–1298. https://doi.org/10.1080/03067319.2015.1100723.

[36] Bonnechère, A.; Hanot, V.; Bragard, C.; Bedoret, T.; van Loco, J. (2012). Effect of household and industrial processing on the levels of pesticide residues and degradation products in melons. *Food Additives & Contaminants: Part A*, *29*(7), 1058–1066. https://doi.org/10.1080/19440049.2012.672339.

[37] Hussain, S.; Siddique, T.; Arshad, M.; Saleem, M. (2009). Bioremediation and phytoremediation of pesticides: Recent advances. *Critical Reviews in Environmental Science and Technology*, *39*(10), 843–907. https://doi.org/10.1080/10643380801910090.

[38] Chelme-Ayala, P.; El-Din, M.G.; Smith, D.W. (2010). Treatability study on membrane concentrate containing pesticides using advanced oxidation processes. *Ozone: Science & Engineering*, *32*(1), 16–24. https://doi.org/10.1080/01919510903468029.

[39] Dong, X.; Lan, T.; Tian, X.; Li, Y.; Zhao, Y.; Zong, Q.; Liu, S.; Pan, C. (2021). Simultaneous determination of 14 pesticide residues in tea by multi-plug filtration cleanup combined with LC-MS/MS. *Journal of Environmental Science and Health, Part B*, *56*(8), 771–781. https://doi.org/10.1080/03601234.2021.1944962.

[40] Magga, Z.; Tzovolou, D.N.; Theodoropoulou, M.A.; Dalkarani, T.; Pikios, K.; Tsakiroglou, C.D. (2008). Soil column experiments used as a means to assess transport, sorption, and biodegradation of pesticides in groundwater. *Journal of Environmental Science and Health, Part B, Pesticides, Food Contaminants, and Agricultural Wastes*, *43*(8), 732–741. https://doi.org/10.1080/03601230802388868.

[41] Lees, K.E.; Fitzsimons, M.F.; Snape, J.; Tappin, A.; Comber, S.D.W. (2020). Developing the OECD 106 fate testing protocol for active pharmaceuticals in soil. *Environmental Technology*, pp. 1–11. https://doi.org/10.1080/09593330.2019.1706643.

[42] Liu, J.; Dai, J.; Wang, R.; Li, F.; Du, X.; Wang, W. (2010). Adsorption/desorption and fate of Mercury (II) by typical black soil and red soil in China. *Soil and Sediment Contamination: An International Journal*, *19*(5), 587–601. https://doi.org/10.1080/15320383.2010.499925.

[43] Muhire, J.; Li, Sha Sha; Yin, B.; Mi, Jia Ying; Zhai, Hong Lin. (2021). A simple approach to the prediction of soil sorption of organophosphorus pesticides. *Journal of Environmental Science and Health, Part B, Pesticides, Food Contaminants, and Agricultural Wastes*, *56*(6), 606–612. https://doi.org/10.1080/03601234.2021.1934358.

[44] Pérez-Lucas, G.; el Aatik, A.; Vela, N.; Fenoll, J.; Navarro, S. (2020). Exogenous organic matter as strategy to reduce pesticide leaching through the soil. *Journal of Environmental Science and Health, Part B*, *67*(7), 934–945. https://doi.org/10.1080/03650340.2020.1768531.

[45] Imache, A.E.; Dousset, S.; Satrallah, A.; Dahchour, A. (2012). Effects of sewage sludge amendments on pesticide sorption and leaching through undisturbed Mediterranean soils. *Journal of Environmental Science and Health, Part B*, *47*(3), 161–167. https://doi.org/10.1080/03601234.2012.632260.

[46] Yadav, S.; Banerjee, T.; Singh, N. (2021). Leaching behaviour of atrazine and fipronil in sugarcane trash ash mixed soils. *International Journal of Environmental Analytical Chemistry*, *103*(19), 7494–7504. https://doi.org/10.1080/03067319.2021.1972101.

[47] Bajeer, M.A.; Mallah, M.A.; Sherazi, S.T.H.; Bhanger, M.I.; Nizamani, S.M. (2015). Investigation of dissipation, adsorption, degradation, and leaching of triazophos pesticide in various soils. *Polycyclic Aromatic Compounds*, *36*(3), 229–241. https://doi.org/10.1080/10406638.2014.964424.

[48] Felsot, A. S.; Unsworth, J. B.; Linders, J. B. H. J.; Roberts, G.; Rautman, D.; Harris, C.; Carazo, E. (2010). Agrochemical spray drift; assessment and mitigation—A review. *Journal of Environmental Science and Health, Part B*, *46*(1), 1–23. https://doi.org/10.1080/03601234.2010.515161.

[49] Langenbach, T.; Mano, D.; Campos, M.M.; Cunha, A.L.M.C.; De Campos, T.M.P. (2017). Pesticide dispersion by spraying under tropical conditions. *Journal of Environmental Science and Health. Part B, Pesticides, Food Contaminants, and Agricultural Wastes*, *52*(12), 843–849. https://doi.org/10.1080/03601234.2017.1359040.

[50] Jensen, A.R.; Spliid, N.H.; Svensmark, B. (2007). Determination of volatilization (dissipation) and secondary deposition of pesticides in a field study using passive dosimeters. *International Journal of Environmental Analytical Chemistry*, *87*(13–14), 913–926. https://doi.org/10.1080/03067310701455955.

[51] Yigit, N.; Velioglu, Y.S. (2019). Effects of processing and storage on pesticide residues in foods. *Critical Reviews in Food Science and Nutrition*, *60*(21), 1–20. https://doi.org/10.1080/10408398.2019.1702501.

[52] Daouk, S.; De Alencastro, L.F.; Pfeifer, H.-R. (2013). The herbicide glyphosate and its metabolite AMPA in the Lavaux vineyard area, western Switzerland: Proof of widespread export to surface waters. Part II: The role of infiltration and surface runoff. *Journal of Environmental Science and Health, Part B*, *48*(9), 725–736. https://doi.org/10.1080/03601234.2013.780548.

[53] Tiktak, A.; Boesten, J.J.T.I.; Egsmose, M.; Gardi, C.; Klein, M.; Vanderborght, J. (2013). European scenarios for exposure of soil organisms to pesticides. *Journal of Environmental Science and Health, Part B, Pesticides, Food Contaminants, and Agricultural Wastes*, *48*(9), 703–716. https://doi.org/10.1080/03601234.2013.780525.

[54] Dołowy, M.; Miszczyk, M.; Pyka, A. (2014). Application of various methods to determine the lipophilicity parameters of the selected urea pesticides as predictors of their bioaccumulation. *Journal of Environmental Science and Health, Part B*, *49*(10), 730–737. https://doi.org/10.1080/03601234.2014.929481.

[55] Reponen, P.; Abass, K.; Mattila, S.; Pelkonen, O. (2010). Overview of the metabolism and interactions of pesticides in hepaticin vitrosystems. *International Journal of Environmental Analytical Chemistry*, *90*(3–6), 429–437. https://doi.org/10.1080/03067310903194931.

[56] Kozawa, K.; Aoyama, Y.; Mashimo, S.; Kimura, H. (2009). Toxicity and actual regulation of organophosphate pesticides. *Toxin Reviews*, *28*(4), 245–254. https://doi.org/10.3109/15569540903297808.

[57] Arcury, T.A.; Quandt, S.A. (2006). Health and social impacts of tobacco production. *Journal of Agromedicine*, *11*(3–4), 71–81. https://doi.org/10.1300/J096v11n03_08.

[58] Frazier, L.M. (2007). Reproductive disorders associated with pesticide exposure. *Journal of Agromedicine*, *12*(1), 27–37. https://doi.org/10.1300/j096v12n01_04.

[59] Sahu, P.; Michael, H.A.; Voss, C.I.; Sikdar, P.K. (2013). Impacts on groundwater recharge areas of megacity pumping: Analysis of potential contamination of Kolkata, India, water supply. *Hydrological Sciences Journal*, *58*(6), 1340–1360. https://doi.org/10.1080/02626667.2013.813946.

[60] Kazemzadeh-Parsi, M.J.; Daneshmand, F.; Ahmadfard, M.A.; Adamowski, J.; Martel, R. (2014). Optimal groundwater remediation design of pump and treat systems via a simulation–optimization approach and firefly algorithm. *Engineering Optimization*, *47*(1), 1–17. https://doi.org/10.1080/0305215x.2013.858138.

[61] Zhou, M.; Cai, F.; Uenishi, M.; Sekine, Y. (2021). Performance evaluation of a hybrid heating system combined with a groundwater source heat pump with an existing fuel oil heater for a horticultural greenhouse. *International Journal of Green Energy*, *19*(13), 1404–1414. https://doi.org/10.1080/15435075.2021.2000415.

[62] Sharief, V.; Eldho, T.I.; Ish, F.; Rastogi, A.K. (2008). Optimal pumping policy for aquifer decontamination by pump and treat method using genetic algorithm. *ISH Journal of Hydraulic Engineering*, *14*(2), 1–17. https://doi.org/10.1080/09715010.2008.10514901.

[63] Hata, T.; Miyata, Y.; Honjo, Y. (2010). Pumping-rate control for contaminated groundwater with VOCs by using fuzzy inference model. *Georisk: Assessment and Management of Risk for Engineered Systems and Geohazards*, *4*(2), 63–76. https://doi.org/10.1080/17499510903045979.

[64] Tsitonaki, A.; Petri, B.; Crimi, M.; Mosbæk, H.; Siegrist, R.L.; Bjerg, P.L. (2010). In situ chemical oxidation of contaminated soil and groundwater using persulfate: A review. *Critical Reviews in Environmental Science and Technology*, *40*(1), 55–91. https://doi.org/10.1080/10643380802039303.

[65] Tressler, A.; Uchrin, C. (2014). Mathematical simulation of chlorinated ethene concentration rebound after in situ chemical oxidation. *Journal of Environmental Science and Health, Part A*, *49*(8), 869–881. https://doi.org/10.1080/10934529.2014.893790.

[66] Wang, P.; Li, J.; An, P.; Yang, B.; Hou, D.; Pu, S. (2023). Understanding the dilemmas and breakdown of the reactive migration of in situ groundwater injection reagents from an

environmental geology perspective. *Critical Reviews in Environmental Science and Technology*, 1–24. https://doi.org/10.1080/10643389.2023.2277649.

[67] Qian, Y.; Wang, Q.; Yue, F. (2014). Remediation of TCE-contaminated water by enhanced chemical oxidation using Na2S2O8/H2O2/red mud. *Desalination and Water Treatment*, *57*(9), 4154–4161. https://doi.org/10.1080/19443994.2014.988648.

[68] Ikehata, K.; Wang-Staley, L.; Qu, X.; Li, Y. (2016). Treatment of groundwater contaminated with 1,4-dioxane, tetrahydrofuran, and chlorinated volatile organic compounds using advanced oxidation processes. *Ozone: Science & Engineering*, *38*(6), 413–424. https://doi.org/10.1080/01919512.2016.1198686.

[69] Kunukcu, Y.K. (2007). In situ bioremediation of groundwater contaminated with petroleum constituents using oxygen release compounds (ORCs). *Journal of Environmental Science and Health, Part A*, *42*(7), 839–845. https://doi.org/10.1080/10934520701373174.

[70] Ye, S.; Zeng, G.; Wu, H.; Zhang, C.; Dai, J.; Liang, J.; Yu, J.; Ren, X.; Yi, H.; Cheng, M.; Zhang, C. (2017). Biological technologies for the remediation of co-contaminated soil. *Critical Reviews in Biotechnology*, *37*(8), 1062–1076. https://doi.org/10.1080/07388551.2017.1304357.

[71] Low, A.; Schleheck, D.; Khou, M.; Aagaard, V.; Lee, M.; Manefield, M. (2007). Options for in situ remediation of soil contaminated with a mixture of perchlorinated compounds. *Bioremediation Journal*, *11*(3), 113–124. https://doi.org/10.1080/10889860701548556.

[72] Kamarudheen, N.; Chacko, S.P.; George, C.A.; Somachandran, R.C.; Rao, B. (2020). An ex-situ and in vitro approach towards the bioremediation of carcinogenic hexavalent chromium. *Preparative Biochemistry & Biotechnology*, *50*(8), 842–848. https://doi.org/10.1080/10826068.2020.1755868.

[73] Bhowmik, A.; Asahino, A.; Shiraki, T.; Nakamura, K.; Takamizawa, K. (2009). In situ study of tetrachloroethylene bioremediation with different microbial community shifting. *Environmental Technology*, *30*(14), 1607–1614. https://doi.org/10.1080/09593330903369986.

[74] Compernolle, T.; Van Passel, S.; Weyens, N.; Vangronsveld, J.; Lebbe, L.; Thewys, T. (2012). Groundwater remediation and the cost effectiveness of phytoremediation. *International Journal of Phytoremediation*, *14*(9), 861–877. https://doi.org/10.1080/15226514.2011.628879.

[75] Weishaar, J.A.; Tsao, D.; Burken, J.G. (2009). Phytoremediation of BTEX hydrocarbons: Potential impacts of diurnal groundwater fluctuation on microbial degradation. *International Journal of Phytoremediation*, *11*(5), 509–523. https://doi.org/10.1080/15226510802656326.

[76] Lafleur, B.; Sauvé, S.; Duy, S.V.; Labrecque, M. (2016). Phytoremediation of groundwater contaminated with pesticides using short-rotation willow crops: A case study of an apple orchard. *International Journal of Phytoremediation*, *18*(11), 1128–1135. https://doi.org/10.1080/15226514.2016.1186593.

[77] Khalid, S.; Shahid, M.; Dumat, C.; Niazi, N. K.; Bibi, I.; Gul Bakhat, H. F. S.; Abbas, G.; Murtaza, B.; Javeed, H. M. R. (2017). Influence of groundwater and wastewater irrigation on lead accumulation in soil and vegetables: Implications for health risk assessment and phytoremediation. *International Journal of Phytoremediation*, *19*(11), 1037–1046. https://doi.org/10.1080/15226514.2017.1319330.