1. Introduction

The ever-increasing anthropogenic activities around the world have resulted in several areas being severely contaminated with hazardous contaminants such as pesticides and heavy metals [1,2]. Pesticides are widely used to control pests and are possibly the most extensively distributed contaminants in the environment [3]. There are countless long-term contaminated sites with great levels of pesticides due to the dumping of obsolete pesticide stocks. Additionally, pollution caused by heavy metals is also among the top most relevant environmental problems today [4]. The extensive heavy metal usage in numerous practices has led to their global spread in silt, soil, waste, and wastewater. Toxic metals polluting the environment came about from several human activities, mainly industrial activities, although agriculture, municipal landfill, and sewage disposal activities had similarly contributed to the pollution [4,5]. Considering these problems, eco-friendly methods to clean up polluted environments by means of various microbial species have emerged to be the preferred pollutant clean-up approach. This approach is more acceptable as it is less intrusive and more remedial of soil functions in comparison to standard physicochemical approaches [6]. Moreover, biological approaches have better flexibility as they involve living systems, hence having the ability to perform complex reactions such as degrading organic pollutants and converting inorganic compounds into nontoxic products [7,8].

Contaminated areas naturally contain microorganisms having the ability to degrade an extensive range of contaminants. However, several pollutants exhibit resistance to biodegradation due to various factors such as low water solubility, low biodegradability, low bioavailability, high toxicity, and high stability [9]. Besides, certain compounds may be inefficiently used as substrates for microbial metabolism. Due to the complexity of the chemical structure of some compounds, a microbial consortium may be required for them to undergo biodegradation. Recalcitrant compounds (slowly biodegradable or non-biodegradable compounds) can be new in many cases, and therefore microorganisms are unlikely to be adapted in using them as substrates [10]. Bioaugmentation is a suitable approach to overcome these problems as its treatment can be modified to the targeted pollutant dominantly present in the environment. It can target not only pollutants in increasing...
amongst environmental microorganisms, actinobacteria have been broadly reported as probable bioremediation agents [3,11]. In nature, actinobacteria are dominant colonizers [12] and are important for recycling refractory biomaterials through humus formation and decomposition [13]. Due to their beneficial functions in nature, actinobacteria have received notable attention, especially on their capability to degrade organic and inorganic pollutants. In this review, a compilation of successful usage of actinobacteria in bioaugmentation within 15 years is described with many case studies. Although, there are many other contaminants present worldwide, this review focuses on pesticides and heavy metals as these two are the most prevalent contaminants in the environment.

2. General concept of bioaugmentation

In general, bioremediation refers to the method of using microorganisms, plants or their enzymes to reduce or remove environmental contaminants or toxic substances. The detoxification can be achieved through different methods including the addition of nutrient to the affected area (biostimulation or biorestoration) and the addition of microbe with specific characteristic (bioaugmentation) [14]. Therefore, it is important to understand the principal and differentiate all bioremediation methods to achieve the main goal; to reduce the concentration or level of toxic contaminants to acceptable level following the environmental requirement or to completely remove it [15].

Bioaugmentation is a type of bioremediation method whereby environmental samples such as soil, sediments and water are inoculated with microorganisms characterized with the preferred catalytic abilities to accelerate the degradation of contaminants (Fig. 1a) [16], [17], [18]). It is mainly carried out in areas where there is low number of autochthonous microorganisms with the ability to degrade the contaminant, and/or the necessary catabolic pathways needed to metabolize pollutants is unavailable in the native population [16,17]. In selecting the appropriate strains for bioaugmentation, the following features of the microorganisms need to be considered: good synergistic interactions among introduced and indigenous microbes, high capacity to degrade contaminants, ease of cultivation, fast growth, endurance in high concentrations of pollutants, and survival in various environmental conditions [17,19].

Based on the origin of the inoculants, bioaugmentation has three different approaches (Fig. 1b). In allochthonous bioaugmentation, microorganisms taken from another site are introduced into the contaminated site, whereas in autochthonous bioaugmentation, microorganisms originally from the polluted site are re-injected back into the same polluted environment. In gene bioaugmentation, genetically engineered microorganisms (GEMs) equipped with genes encoding for enzymes responsible for specific desired functions are introduced directly into the environment. This is to enhance the rate of contaminant degradation by increasing, through horizontal gene transfer, the number and diversity of indigenous microorganisms to metabolize the target contaminants [18], [20–22]). Examples of plasmid that have been reported in plasmid-mediated bioaugmentation includes pJP4 for 2, 4-dichlorophenoxyacetic acid (2,4-D) degradation [23], pDOC for chlorpyrifos degradation [24], and pDOD for dichlorodiphenyltrichloroethane (DDT) degradation [20]. Degradation of these pesticides are important as they are widely used in agriculture but have caused serious toxicological and environmental problems.

As such, there has been an increasing interest in using Actinobacteria to remove toxic substances and soil contaminants via bioaugmentation. Through metagenomic analysis done by various researchers, Actinobacteria is either the second or third major groups found in contaminated soils with heavy metals [25–27], and pesticides [28,29]. This proves that Actinobacteria is a good candidate to be used in bioaugmentation.

3. Actinobacteria

Actinobacteria is one of the largest phylum under Bacteria domain and can be found in a wide range of terrestrial and aquatic ecosystems. Actinobacteria are Gram-positive bacteria with > 50% of guanine and cytosine (G + C) content in their DNA. Generally, actinobacteria are recognized as filamentous bacteria due to their ability to form substrate mycelium and aerial mycelium [30]. Until 2016, 374 genera of actinobacteria have been recorded [31]. Actinobacteria can be found either as Streptomyces, which is the dominant genus in the group, or as non-Streptomyces actinomycetes such as the genus of Actinomyces, Kitasatospora, Micromonospora, Nocardia, Micrococcus, Arthrobacter and Rhodococcus [32]. Streptomyces covers a large number of species with great diversity in culture conditions [33]. In contrast, non-Streptomyces actinomycetes are strains with low frequency of isolation under normal conditions and most of them require specific isolation, preservation and cultivation methods [34,35].

The phylum Actinobacteria represents the most recognized group of

![Fig. 1. (a) Bioaugmentation method; (b) Three different approaches of bioaugmentation.](image-url)
microorganisms with the ability to produce bioactive compounds. Thus, actinobacteria have received great interest in various applications in pharmaceuticals, biotechnology, food industries, agriculture and in the enzyme industry [36-38]. Moreover, there is also interest to use Actinobacteria to assist in pollutant clean-up. Actinobacteria provides an economic and safe biological method for removal of contaminants as they can metabolize the contaminants for their growth [39].

4. Actinobacteria with pesticide degrading abilities

Over the years, there have been many research findings on pesticide degradation by actinobacteria. It has been noted that Streptomyces and non-Streptomyces actinomycetes of the genus Actinoplanes, Frankia, Gordonia, Kocuria, Nocardioidea and Rhodococcus are among the commonly reported pesticide degrading actinobacteria. These microorganisms exhibit abilities to grow, metabolize and degrade several chemical families of pesticides, including organophosphorous (chlorpyrifos, dimethoate), organochlorine (endosulfan, lindane, pentachlorophenol, pentachloronitrobenzene, hexachlorobenzene), triazine (atrazine, simazine, terbutylazine), synthetic pyrethroids (deltamethrin, cypermethrin) and benzimidazole (carbendazim) compounds [3, 40]. Examples of actinobacteria with pesticide degrading abilities are listed in Table 1.

5. Bioaugmentation of pesticide-contaminated soils using pesticide degrading actinobacteria

Although various studies have been conducted to isolate and characterize the degrading capacity of pesticide degrading actinobacteria, and with some studies showing bioaugmentation potentials [45, 51, 56, 70, 68], not all pesticide degrading actinobacteria have been studied further on their potentials to be used for the bioaugmentation of pesticide-contaminated soils. Table 2 describes and summarizes studies that have reported on the use of pesticide degrading actinobacteria to decontaminate pesticides present in soil via bioaugmentation.

Endosulfan is an insecticide that has been used extensively worldwide to control pests on crops. It is a mixture of two stereoisomers, α- and β-endosulfan, in a ratio of 7:3. Endosulfan is toxic, bioaccumulative and persistent in the environment [41, 71]. Kumar et al. (2008) reported on the bioaugmentation of endosulfan-contaminated soil with Arthrobacter sp. [41]. Through soil microcosm studies, inoculation of Arthrobacter sp. (2 × 10^7 CFU/g) into endosulfan-contaminated soil resulted in 73% and 75% degradation of α- and β-endosulfan (50 mg/kg), respectively after 6 weeks. As for the uninoculated soils, degradation of α- and β-endosulfan after 6 weeks was 26% and 23%, respectively [41].

Atrazine is one of the most widely used and toxic herbicides to control weeds in crops. It is a pollutant of great concern due to its high potential to cause soil and water contamination [42,72]. The genus Arthrobacter has been recognised as capable of degrading many types of xenobiotics [3]. An atrazine degrading Arthrobacter sp. strain DAT1 was reported to be able to utilize atrazine as a sole nitrogen source for growth and harboured atrazine-degrading genes (atZB, atZC and trzN) [42]. Wang et al. (2013) later investigated the potential of strain DAT1 in remediating heavily atrazine-contaminated soil [73]. Strain DAT1 demonstrated highly effective removal of atrazine (400 mg/kg) from contaminated soil, whereby the inoculation of strain DAT1 (1.3 × 10^7 CFU/g) into both autoclaved and non-autoclaved soils had resulted in a nearly complete removal of atrazine (>95%) in 3 days. Moreover, results on the inoculants durability/ survival and catabolic potential for soil remediation using terminal restriction fragment length polymorphism (TRFLP) and quantitative PCR (qPCR) targeting the atrazine-degrading genes demonstrated a steady increase of the tested genes and the relative abundance of strain DAT1 in the bioaugmented

Table 1

| Actinobacteria | Pesticide degraded | Source of isolation | Ref |
|---------------|--------------------|---------------------|-----|
| Arthrobacter sp. | α-endosulfan, β-endosulfan | Soil from different agricultural fields contaminated with pesticides, India | [41] |
| Arthrobacter sp. strain DAT1 | Atrazine | Atrazine-contaminated agricultural soil in Hebei Province, China | [42] |
| Arthrobacter sp. strain AK-YN10 | Atrazine | Soil from atrazine-contaminated agricultural field in Maharashtra, India | [63, 44] |
| Arthrobacter sp. C3 | Atrazine | Soil from atrazine-sprayed corn field in China | [45] |
| Arthrobacter sp. SD5-25 | Simazine | Soil from simazine-terminated farmland in Liaoning Province, China | [46] |
| Arthrobacter aureus TCI | Terbutylazine | Soil from atrazine-contaminated spill site in South Dakota, USA | [47, 48] |
| Frankia alni ACN1| Atrazine | Not reported | [49] |
| Frankia sp. Eul1c | Atrazine | Not reported | [49] |
| Gordonia sp. JAA51 | Chlorpyrifos | Paddy field soil exposed to chronic chlorpyrifos applications in Tamil Nadu, India | [50] |
| Kocuria sp. CL2 | Pentachlorophenol | Sludge of pulp and paper mill | [51] |
| Kocuria sp. DAB-1Y | Lindane | Soil from a contaminated site in Lucknow, India with history of lindane application | [52] |
| Kocuria kristinae | Chlorpyrifos | Chlorpyrifos-contaminated agricultural soil in India | [53] |
| Kocuria turfanensis | Dimethoate | Soil from long history of pesticide application in India | [54] |
| Nocardioidea sp. strain PD653 | Pentachloronitrobenzene, Hexachlorobenzene, Pentachlorophenol | Soil from an agricultural field contaminated with pentachloronitrobenzene in Japan | [55] |
| Nocardioidea sp. strain SG-4G | Carbendazim | Carbendazim-exposed soil in Australia | [56] |
| Nocardioidea soli | Carbendazim | Soil under long-term applications of carbendazim in Changzhou, China | [57] |
| Rhodococcus sp. BCH2 | Atrazine | Soil from grape farm with long-term atrazine treatment near Sangli, India | [58] |
| Rhodococcus sp. strain MB-P1 | Atrazine | Atrazine-contaminated soil in India | [59] |
| Rhodococcus sp. djl-6 | Carbendazim | Carbendazim-treated soil of a vegetable field in Taixing, China | [60] |
| Rhodococcus erythropolis djl-11 | Carbendazim | Soil from vineyards in Rhizao, China with 10-year history of carbendazim application | [61] |
| Rhodococcus julianiae djl-6-2 | Carbendazim | Sludge of a carbendazim wastewater treatment facility in Jiangsu province, China | [62] |
| Rhodococcus qinghengii djl-6 | Carbendazim | Carbendazim-contaminated soil from a vegetable field in Jiangsu province, China | [63] |
| Streptomyces aureus HP-S-01 | Deltamethrin, Cypermethrin | Activated sludge from a pyrethroid manufacturing wastewater treatment system in Zhongshan, China | [64, 65] |
| Streptomyces sp. M7 | α-HCH, β-HCH, Lindane (γ-HCH) | Wastewater sediment of a copper filter plant in Tucuman, Argentina | [66, 67, 68] |
| Streptomyces sp. strains A2, A5, A11 | Lindane | Organochlorine pesticide-contaminated soil in Santiago del Estero, Argentina | [69] |
Characteristics of pesticide degrading actinobacteria used in the bioaugmentation of pesticide contaminated soils.

| Actinobacteria | Pesticide degraded | Dosage (mg/kg) | Inoculum size (CFU/g) | Removal efficiency/ comments | Ref |
|----------------|--------------------|----------------|-----------------------|-----------------------------|-----|
| Arthrobacter sp. | α-endo-sulfan, β-endo-sulfan | 50 | 2 × 10^8 CFU/g | Degradation of 73% α-endo-sulfan and 75% β-endo-sulfan after 6 weeks | [41] |
| Arthrobacter sp. strain DAT1 | Atrazine | 400 | 1.3 × 10^8 CFU/g | >95% of atrazine removal in 3 days | [73] |
| Arthrobacter sp. strain AK-YN10 | Atrazine | 100 | 2.5 × 10^8 CFU/g per 250 g soil | Complete removal of atrazine within 12 days | [44] |
| Arthrobacter sp. SD3-25 | Simazine | 200 | 3.3 × 10^8 CFU/g | >90% of simazine removal within 7 days | [46] |
| Arthrobacter aureus TCI | Terbutylazine | 3.8 | 5 × 10^7 - 2.0 × 10^8 CFU/g | 95% of terbutylazine removal within 3 days | [48] |
| Rhodococcus jialiangiae djl-6-2 | Carbendazim | 10 | 1 × 10^6 CFU/g | Almost 100% carbendazim removal after 20 days | [77] |
| Rhodococcus qingshengi djl-6 | Carbendazim | 2, 8 | 1 × 10^6 CFU/g | > 93% of carbendazim removal after 14 days of incubation | [76] |
| Streptomyces aureus HP-8-01 | Cypermethrin, 3-PBA | 50 | 1 × 10^6 CFU/g | Removal of 81.1% cypermethrin and 73.0% 3-PBA after 10 days (field experiments) | [65] |
| Streptomyces consortium (strains A2, A5, A11, M7) | Lindane | 35 | 1 × 10^7 CFU/g | Degradation of other pyrethroids such as bifenthrin, cyfluthrin, fenvalerate, fenpropathrin and permethrin | [79] |
| Streptomyces consortium (strains A2, A5, A11, M7) | Lindane | 100 | 1 × 10^8 CFU/g | Lindane removal from concentrated slurry (soil: water ratio of 2:3) within 7 days | [80] |
| Streptomyces consortium (strains A2, A5, A11, M7) | Lindane | 2 | 2 g/kg, wet weight | Microcosms studies: 61–86% of lindane removal in bioaugmented and biostimulated soils after 14 days | [81] |

The use of fungicide carbendazim to control fungal diseases in agriculture had led to its accumulation and environmental contamination worldwide. Furthermore, carbendazim undergoes slow degradation and can persist in bare soil for a long time [75,76]. The genus Rhodococcus can utilize recalcitrant and toxic compounds for its growth and metabolism, and majority of carbendazim-degrading bacteria belongs to this genus [75]. The carbendazim-degrading Rhodococcus jialiangiae djl-6-2 was reported to be capable of utilizing carbendazim as the sole carbon and nitrogen source for growth [62,77].
carbendazim on bacterial community composition and reduced the fungicidal activity of carbendazim. Under strain-inoculated conditions, more than 93% of carbendazim was removed from both low dose (2 mg/kg) and high dose (8 mg/kg) carbendazim-treated soils after 14 days of incubation, as compared to 29% of carbendazim removal from uninoculated soils [76].

Synthetic pyrethroids insecticides are used to control insect pests in both agriculture and home. Although potently neurotoxic against insects and low toxicities for mammals, it is potentially harmful to human health and the environment [64,65]. The deltamethrin-degrading Streptomyces aureus HP-S-01 was reported to be highly effective in degrading pyrethroids such as deltamethrin, bifenthrin, cyfluthrin, cypermethrin, fenvalerate, fenpropatrin and permethrin, and also their β-cypermethrin, fenpropathrin and permethrin, and also their Streptomyces aureus health and the environment [64,65]. Chen et al. (2012) had conducted laboratory and field-scale experiment to investigate the ability of inoculated strain HP-S-01 to eliminate β-cypermethrin and 3-PBA in soils [65]. After introduction with strain HP-S-01 (1 × 10⁹ CFU/g), laboratory results showed that the removal rates of the initial doses of β-cypermethrin and 3-PBA (50 mg/kg) were higher in non-sterilized soils (87.8% and 79.3%, respectively) when compared to sterilized soils (80.5% and 73.1%, respectively). This indicated that soil microorganisms may have enhanced the ability of strain HP-S-01 in removing β-cypermethrin and 3-PBA. Enhanced removal may be due to the synergistic abilities between strain HP-S-01 and the indigenous soil microbial community in degrading β-cypermethrin and 3-PBA. Similar results were observed in field experiments whereby strain HP-S-01 had removed 81.1% and 73.5% of initial concentrations of β-cypermethrin and 3-PBA, respectively from the soil after 10 days [65].

Lindane or γ-hexachlorocyclohexane (γ-HCH) is a toxic insecticide that has been extensively used for agriculture. Lindane residues can leak through soil surfaces to ground water, resulting pollution of aquatic ecosystems [78]. Saez et al. (2014) demonstrated efficient lindane removal from concentrated soil slurry by a Streptomyces consortium (consisting of Streptomyces strains A2, A5, A11, M7) immobilized in cloth sachets [79]. From testing various inoculum sizes, lindane, and slurry concentration, best lindane removal (35 mg/kg of soil) from concentrated slurry (soil: water ratio of 2:3) was achieved within 7 days of incubation using Streptomyces inoculum of 1 × 10⁹ CFU/g. As for diluted slurry (soil: water ratio of 1:4), 28.7 mg/kg of soil of lindane was removed by the immobilized Streptomyces consortium after 14 days [79]. Later, Saez et al. (2018) investigated the bioaugmentation effect of a mixed culture consisting of the Streptomyces consortium (strains A2, A5, A11, M7) and fungi, on their capacity to remove lindane (100 mg/kg) from biomixture samples formulated with clayey soil, sandy soil, and silty loam soil [80]. It was observed that at 60 days of incubation, lindane removal achieved by the bioaugmented biomixtures were about 81–87% while the non-bioaugmented mixtures only showed 55–70% of lindane removal [80].

Moreover, Raimondo et al. (2020a) reported microcosm studies involving the applications of bioaugmentation with Streptomyces consortium (Streptomyces strains A2, A5, A11, M7) and biostimulation with sugarcane filter cake to biomediate lindane-contaminated soils [81]. Inoculation with the Streptomyces consortium (2 g/kg, wet weight) and biostimulation resulted in lindane (2 mg/kg) removal of 70.8%, 61.4% and 86.3% from clayey soil, silty loam soil and sandy soil, respectively after 14 days. In contrast, non-bioaugmented and non-biostimulated microcosms only showed about 26–35% of lindane removal [81]. Later, in another study, Raimondo et al. (2020b) reported successful scaling-up from microcosms to mesocosms for the combined applications of bioaugmentation with Streptomyces consortium and biostimulation with sugarcane filter cake for the bioremediation of lindane-contaminated soils [82]. After 63 days of incubation, inoculation with the Streptomyces consortium (2 g/kg, wet weight) and biostimulation showed lindane (2 mg/kg) removal of 68.6%, 80% and 82.6% from clayey soil, silty loam soil and sandy soil, respectively. For non-bioaugmented and non-biostimulated mesocosms, only about 32–38% of lindane removal observed [82].

6. Actinobacteria with heavy metal resistance and capabilities in heavy metal removal

Heavy metals are metallic elements found in natural sources (soil, sediments, water, air and living organisms), and also from anthropogenic sources. Heavy metals of anthropogenic origin generate a continuous and everlasting pollution and are usually related to agriculture, industrial and urbanization sectors [2,83]. Actinobacteria have been found to be either the second or third major groups in contaminated soils [25–27] and therefore, is a potential group that can be used for cleaning heavy metal pollution together with the indigenous microorganisms in soil. Besides that, major probiotics such as Bifidobacteria, found in human intestinal tract have also been proven to be able to remove heavy metals, especially from water samples [84,85]. Over the years, several Streptomyces and non-Streptomyces actinomycetes of the genus Acinetobacter, Arthrobacter, Bifidobacterium, Micrococcus, Nocardioides, Nocardia, Propionibacterium and Rhodococcus have been identified to exhibit heavy metal resistance, with some having capabilities in heavy metal removal (Table 3), which indicates their potential capacity as tools for heavy metal bioremediation.

7. Bioaugmentation of heavy metals by actinobacteria

The success of bioaugmentation depends on the survival of the inoculated strains upon introduction to the bioaugmented soil. Hence, in heavy metal contaminated sites, actinobacteria with capabilities to resist or tolerate high concentrations of heavy metals will have better adaptability and survival in these stressed environments, making them suitable to be applied for heavy metal bioremediation. Although various studies have identified heavy metal resistant actinobacteria, and with some showing capabilities in heavy metal removal (Table 3), the potentials of heavy metal resistant actinobacteria for use in bioaugmentation approach have not been much explored. Table 4 lists some actinobacteria that have been studied for heavy metal removal via bioaugmentation.

Amycolatopsis sp. A80 or Amycolatopsis tucumanensis DSM 45,259, isolated from copper polluted sediments [104,105], demonstrated remarkable copper resistance and was able to reduce copper up to 70% in minimal media [106]. Further study on bioaugmentation of copper in loamy soil samples showed that strain DSM 45,259 was able to reduce copper up to 31%. Strain DSM 45,259 was able also to colonize well in the copper contaminated soil microcosms, which confirmed its resistance towards copper in the soil [107].

Emenike et al. (2017) reported on the isolation of nine bacteria from landfill-leasehate contaminated soil including Rhodococcus sp. in Malaysia [108]. In this study, the residual concentrations of lead, copper and aluminum in the soil were significantly reduced after 100 days of treatment by two different consortia, whereby one consortium consisted of all isolated bacteria (LSA) while another consortium consisted of Bacillus sp., Rhodococcus sp. and Lysinibacillus sp. (LSB). The LSB consortium consisting of Rhodococcus sp. was able to reduce high amounts of heavy metal from the leachate contaminated soil, indicating that Rhodococcus sp. is a potential candidate in the combination of bacteria for bioaugmentation that can speed up the removal of heavy metal processes [108].

In a bioaugmentation study done by Strachel et al. (2020), two out of four strains isolated from loamy sand contaminated with zinc were non-Streptomyces species, Gordonia umicola KM113329.1 and Leifsonia sp. KJ191763.1 [109]. It was observed that soil inoculation with the zinc-resistant microbial consortium was able to minimize the adverse effects of zinc contamination in soil. However, unlike other bioaugmentation studies, the microbial consortium in this study was unable to reduce the bioavailable zinc in the loamy sand sample [109].
Actinobacteria that exhibits heavy metal resistance and with capabilities in heavy metal removal.

| Actinobacteria | Heavy metal resistance | Additional info | Ref |
|----------------|------------------------|-----------------|-----|
| Actinobacteria TY046-21, Actinobacteria TY046-078, Actinobacteria TY046-017, | Arsenic, lead, nickel, mercuric ion | Isolated from tin tailings and forest soil | [86] |
| Actinobacter haemolyticus | Chromium | Only strain TY046-017 show tolerance to mercuric ion | [87] |
| Arthrobacter ramosus G2 | Cadmium, cobalt, zinc, chromium, mercury | Isolated from mercuric salt-contaminated soil | [88] |
| Bifidobacterium angulatum | Cadmium, lead, arsenic | Effective removal of cadmium, lead and arsenic in artificially contaminated water | [89] |
| Bifidobacterium longum 46, Bifidobacterium lactis Bb12 | Cadmium, lead | Able to bind and remove cadmium and lead efficiently and rapidly in water | [90] |
| Propionibacterium freudenreichii shermanii JS | Lead | Removal of 56.5% of lead on wastewater | [91] |
| Micrococcus sp. AL06Ni and AL04Co | Nickel, cobalt | Isolated from a heavy metal contaminated soil | [92] |
| Nocardia strain R25 | Chromium, nickel | Isolated from a heavy metal contaminated soil | [93] |
| Nocardia strain MORS2014 | Iron | Able to tolerate 15 mM nickel | [94] |
| Nocardia sp. MORSY2014 | Chromium, zinc, nickel | Isolated from polluted wastewater from industrial regions | [95] |
| Nocardiopsis sp. MORSY1948 | Zinc, nickel, chromium | Isolated from polluted wastewater from industrial regions | [96] |
| Nocardiopsis strain M13 | Lead | Removal of zinc (67.4%), nickel (60.1%), and chromium (47.4%) by isolate biomass | [97] |
| Rhodococcus sp. | Cadmium, lead, aluminum, iron | Isolated from landfill leachate contaminated soil | [98] |
| Rhodococcus sp. CA60Co, AL05Ni, AL32Gd and AL03Ni | Cadmium, chromium, nickel | Isolated from a heavy metal contaminated soil | [99] |
| Streptomyces sp. F4 | Cadmium | Isolated from polluted soil of a former uranium mine | [100] |
| Streptomyces sp. M46 | Chromium | Able to absorb and complex cadmium | [101] |
| Streptomyces sp. strain C11 | Copper, iron, manganese | 50% removal of chromium after 3 days incubation | [102] |
| Streptomyces sp. strain FM1 and FM2 | Lead | Isolated from soil of industrial wastewater treatment plant | [103] |
| Streptomyces acidiscabies E13 | Nickel | Able to grow at ≥5000 mg/l | [104] |
| Streptomyces griseus NCIM 2030 | Chromate | Completely reduced 25 mg/L of Cr(VI) to Cr(III) within 24 hours | [105] |
| Streptomyces rochei ANH | Chromium, cadmium, lead | Isolated from seacoast sediment in Alexandria, Egypt | [106] |
| Streptomyces matansis BG5, Streptomyces vinaceus CRF2, Streptomyces sp. CRF14, Streptomyces pulcher CRF17 | Chromium | Efficient removal of phytotoxic in tannery wastewater | [107] |
| Streptomyces thermacarboxydas | Chromium | Isolated from saline farmlands of Punjab, Pakistan | [108] |
| Streptomyces sinclairis | Zinc, cadmium | Resistant to chromium at 1000 mg/L | [109] |

Streptomyces sp. R25 was isolated from river sediments which were contaminated with heavy metals [110]. Jezéquel and Lebeau (2008) studied strain R25 for the bioaugmentation of cadmium and concluded that strain R25 was competitive towards indigenous microorganisms and can colonize in the soil [111]. Interestingly, strain R25 was also able to reduce cadmium after 3 weeks of incubation [112].

In another study, Polti et al. (2007) successfully isolated Streptomyces sp. strain MCI from sugarcane and demonstrated its ability to reduce Cr (VI) in liquid minimal medium [95]. Later, Polti et al. (2009) investigated the ability of strain MCI to reduce Cr(VI) [112]. Strain MCI was proven to be able to reduce 50 mg/kg of Cr(VI) almost completely (94%) in the normal condition of soil samples after 7 days without any previous treatment and no additional of substrate onto the soil samples. Moreover, the reduction of Cr(VI) activity by strain MCI was not affected by the indigenous microbial flora in soil samples [112].

Apariçio et al. (2018a) studied three different Streptomyces strains (Streptomyces sp. M7, MC1, A5) and one non-Streptomyces strain, Amycolatopsis tucumanensis DSM 45,259, isolated from different soils and sediments contaminated with heavy metals and organochloride pesticides, and investigated the ability of pure and mixed cultures to remove chromium in minimal media as well as in soil artificially polluted with chromium [113]. Overall, the removal of bioavailable chromium from soil by actinobacteria was found to be lower than in liquid system. Using single cultures, strain MCI showed the maximal removal of chromium in the liquid system up to 94% while in the soil sample, strain M7 was able to remove only 50% of the bioavailable fraction of the metal. In the experiment using mixed cultures, the greatest chromium removal efficiency in liquid was obtained in the consortium consisting all four strains, whereby >90% of chromium was removed in the liquid system while 64.4% of bioavailable chromium removed in the liquid system while 64.4% of bioavailable chromium
was removed in the soil [113].

In another study, Apairicio et al. (2018b) used the same actinobacteria consortium to remove chromium and lindane in environmental co-contaminated soils [114]. The bioaugmentation experiment in six different soils collected showed that the consortium was able to reduce the concentration of chromium until not detectable amounts, in five soil samples after 14 days of incubation at optimal temperature and moisture conditions. Chromium removal from one soil sample was not successful due to the inability of the actinobacteria consortium to grow in the alkaline soil condition (pH 9.72). Furthermore, lindane concentrations in the co-contaminated soil studied showed reduction after inoculations with the consortium, indicating the potential of the four strains consortium as a promising tool for the bioremediation of soils co-contaminated with heavy metals and pesticides [114].

8. Conclusion and future prospects

Due to the continuous development on industrial activities, the spread of contaminants worldwide is unceasing as there are insufficient monitoring on the negative effects of these contaminants on the environment. This had led to various research for methods that can improve the conditions of these polluted environments. As such, bioaugmentation with actinobacteria is a beneficial environmentally friendly approach to help restore the environment. Actinobacteria have been proven to be abundant in contaminated soils and are able to metabolize contaminants for their growth. Furthermore, the potentials of actinobacteria in degrading pesticides and removing heavy metals have been demonstrated by various studies, indicating their potential capacities as tools for contaminant bioaugmentation.

However, the bioaugmentation strategy remains a challenging method for efficient elimination of contaminants as there are still limited information on the inoculated strains introduced into the bioaugmented soil. Bioaugmentation efficiency depends mostly on the degrading activity of the inoculated microorganisms and their ability to survive with indigenous microorganisms. Yet, death of the inoculated strains has been observed after introduction to a site, due to abiotic and biotic stresses, and distribution of the introduced strains were limited in the soil matrix [9,18]. Hence, new strategies are needed to improve the efficiency of bioaugmentation. A possible strategy is through quorum sensing modulation to stimulate biofilm formation of the inoculated and soil indigenous microorganisms. Quorum sensing can be controlled and regulated by adding auto-inducers or inhibitors, which may influence biofilm formation and colonization of microorganisms and their bioaugmentation effect [9,40]. Moreover, molecular techniques such as quantitative PCR, e-GFP-tagging, RT-qPCR and stable isotope probing, can be used to monitor the survival and activity of the inoculated strains introduced into the bioaugmented soil [115]. This will be useful for evaluating the performance of the inoculated strains under in situ conditions and will be helpful to improve bioaugmentation effectiveness.

CRediT authorship contribution statement

Christina-Injan Mawang: Conceptualization, Writing – original draft, Writing – review & editing, Validation, Visualization. Adzzi-Shazleen Azman: Conceptualization, Writing – original draft, Writing – review & editing, Validation, Visualization. Aalina-Sakiinah Mohd Fuad: Writing – review & editing. Mariana Ahamad: Writing – review & editing, Validation.

Declaration of Competing Interest

The authors declare no conflict of interest.

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Table 4

| Actinobacteria | Heavy metal | Additional info | Ref |
|----------------|-------------|-----------------|-----|
| Amycolatospis tucumanensis DSM 45,259 | Copper | Copper removal up to 31% in loamy soil samples | [107] |
| Rhodococcus sp | Lead, copper, aluminum | Significant reduction of heavy metals in soil after 100 days treatment | [108] |
| Streptomyces sp. R25 | Cadmium | Cadmium reduction in soil after 3 weeks of incubation | [111] |
| Streptomyces sp. MC1 | Chromium | 94% reduction of chromium in soil samples after 7 days | [112] |
| Streptomyces strains (M7, MC1, A5) and Amycolatospis tucumanensis DSM 45,259 | Chromium | • For single culture:  
• Strain MC1 removed 94% of chromium in liquid system  
• Strain M7 removed 50% of chromium in soil sample  
• For mixed culture:  
• Removal of >90% of chromium in liquid system and 64.4% of bioavailable chromium in soil | [113] |
| Streptomyces strains (M7, MC1, A5) and Amycolatospis tucumanensis DSM 45,259 | Chromium | Chromium removal until not detectable amounts by the consortium after 14 days of incubation | [114] |
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