

Aerobic Degradation of the Antidiabetic Drug Metformin by *Aminobacter* sp. Strain NyZ550

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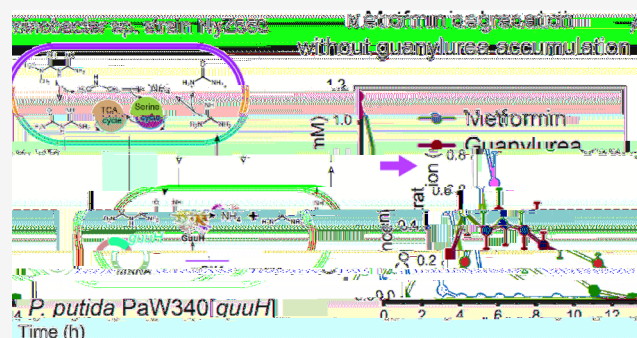


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ABSTRACT: Metformin is becoming one of the most common emerging contaminants in surface and wastewater. Its biodegradation generally leads to the accumulation of guanlyurea in the environment, but the microorganisms and mechanisms involved in this process remain elusive. Here, *Aminobacter* sp. strain NyZ550 was isolated and characterized for its ability to grow on metformin as a sole source of carbon, nitrogen, and energy under oxic conditions. This isolate also assimilated a variety of nitrogenous compounds, including dimethylamine. Hydrolysis of metformin by strain NyZ550 was accompanied by a stoichiometric accumulation of guanlyurea as a dead-end product. Based on ion chromatography, gas chromatography–mass spectrometry, and comparative transcriptomic analyses, dimethylamine was identified as an additional hydrolytic product supporting the growth of the strain. Notably, a microbial mixture consisting of strain NyZ550 and an engineered *Pseudomonas putida* expressing a guanlyurea hydrolase was constructed for complete elimination of metformin and its persistent product guanlyurea. Overall, our results not only provide new insights into the metformin biodegradation pathway, leading to the commonly observed accumulation of guanlyurea in the environment, but also open doors for the complete degradation of the new pollutant metformin.

KEYWORDS: bacteria, biodegradation, emerging contaminant, guanlyurea, metformin



1. INTRODUCTION

Pharmaceuticals in the environment are considered emerging contaminants and are of global concern due to their gradually revealed risks to ecosystems and human health. Pharmaceutical pollution is primarily due to the use of pharmaceuticals by humans or animals, resulting in their continuous release into the environment through wastewater treatment plants worldwide. Most pharmaceuticals are synthetic compounds or natural products with modifications, affecting biological targets even at extremely low concentrations. Their presence in the environment has adverse effects on the ecosystems, aquatic organisms, and humans. However, these effects are largely underestimated due to the lack of comprehensive data on their unpredictable synergistic toxicity and the risks of their structurally diverse products, as well as on the biodegradability and environmental fate of pharmaceuticals.

Metformin (M1), a synthetic guanidine derivative used as a blood glucose-lowering drug for type II diabetes. Currently, more than 500 million people worldwide have diabetes, primarily type II diabetes, with the number estimated to increase to 34 million by 2050. Due to the worldwide prevalence of diabetes and the need for long-term medication, M1 has become one of the most commonly

used prescription drugs. However, M1 is not metabolized by the human body and is excreted as a prototype in urine and feces after oral administration. Therefore, most M1 will converge on rivers, leading to high concentrations of M1 in the wastewater of rivers, with concentrations up to 30. $\mu\text{g L}^{-1}$ observed in untreated effluents. During flow through the rivers, M1 is mainly transformed by biological processes. The present treatment of rivers shows incomplete elimination of M1 with removal rates ranging from 10% to more than 74%, and ultimately significant amount of M1 from rivers is released into the environment. Consequently, the worldwide occurrence of M1 has been observed in surface, ground, drinking and coastal waters, sludges, and soils. It has recently been categorized as one of the top pharmaceuticals present in the environment. M1 exposure exhibits adverse effects on aquatic life and can accumulate in plants. In addition, its transformation

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products such as guanidurea, methylamine, and chlorination byproducts appear to be toxic and widely distributed.

The widespread occurrence and obvious risks of M1 and its derivatives necessitate remediation of M1-contaminated environments. Developing biodegradation technologies compatible with current wastewater treatment processes is considered a cost-effective and promising strategy. However, this is limited by a lack of available M1-degrading bacterial strains and knowledge of their degradative properties. Microbial consortia from soils or activated sewage sludge have been demonstrated to be able to degrade M1 and urea under aerobic or anaerobic conditions.^{7,10,31} It has been suggested that aerobic conditions are preferred for M1 degradation, whereas urea degradation is faster under anaerobic conditions.⁷ However, aerobic degradation of urea was promoted by the addition of glucose as a co-substrate, suggesting that urea can be readily used as a sole nitrogen source.¹⁰ *Pseudomonas mendocina* urea was recently isolated from effluents for its ability to grow on urea as a sole nitrogen source.¹⁰ A novel hydrolase was identified that catalyzed the initial hydrolysis of urea, and the resulting ammonium and guanidine were assimilated by this strain. To date, only a single bacterial strain, *Aminobacter* sp., capable of using M1 as the sole carbon source has been isolated from activated sludge, and the strain converts M1 into the dead-end product urea, but further details of this degradation have not been investigated.¹⁴

In this study, we determined the concentrations of M1 and urea in wastewater from Shanghai and the M1-degrading capacity of microbial consortia from these samples. The results helped us successfully isolate a M1 utilizer, the *Aminobacter* sp. strain NyZ550, capable of growing on M1 as a sole source of carbon, nitrogen, and energy. An investigation of the degradation of M1 by strain NyZ550 revealed the following characteristics: i- strain NyZ550 has a robust ability to degrade M1, ii- strain NyZ550 is a versatile metabolizer of nitrogen-containing compounds, and iii- the degradation of M1 produces urea and dimethylamine, urea via hydrolysis, with the former a dead-end product and the latter supporting the growth of strain NyZ550. Consequently, a microbial mixture composed of strain NyZ550 and an engineered *Pseudomonas putida* urea (expressing urea hydrolase) exhibits a capacity for both M1 and urea degradation.

8. MATERIALS AND METHODS

8.1. Reagents. M1, dimethylbiguanide hydrochloride, guanidurea sulfate salt hydrate, and dicyandiamide were purchased from Shanghai, China. Methylamine, urea, urea (in water, trimethylamine, urea in water, and water-¹⁴) were purchased from Sigma-Aldrich, St. Louis, MO. Guanidine and dimethylguanidine were purchased from AuanAe Bio-Technology, Shanghai, China. Urea, dimethylbiguanide, and biguanide were purchased from Mediatech, Shanghai, China. All chemicals had a purity of >99%. Other commercial inorganic chemicals of analytical grade were used in this study. Ultrapure water and HPLC grade methanol were used to prepare the samples and mobile phases for ultra-performance liquid chromatography (UPLC).

8.2. Wastewater Sample Collection. Wastewater samples were collected from domestic effluents from two cities in China in April 2010. The effluents had a treatment capacity of 45,000–100,000 m³ per day, and all of them had an aeration

tank where the wastewater was sampled. Biodegradability tests were conducted within 2 weeks after the samples were transported to the laboratory. Cor 82–M: analyses, 10 mL of each sample was pretreated by direct filtration with 0.1 μm nylon filters. Filtered samples were adjusted to pH 3.0 with 1 M NaOH or H2I and stored at 4 °C.

8.3. Batch Cultures. Batch experiments to test the biodegradability of the wastewater samples toward M1 were performed in 150 mL Erlenmeyer flasks sealed with the polytetrafluoroethylene ventilatory film, pore diameter 0.1–0.2 μm. The flasks containing 100 mL of minimal salt medium (M: M, pH 3.0) and 5 mM M1 were added with an appropriate volume of the wastewater sample as inoculum to a density of approximately 10⁶ to 10⁷ colony forming units (CFUs) per mL, determined by serial dilution of wastewater and plate count of microorganisms grown on the tryptic soy broth agar plate. The cultures were incubated at 30 °C in a shaking incubator at 150 rpm. Samples were taken at appropriate intervals in triplicate for measuring the concentrations of M1 in the cultures. The cultures were transferred to new flasks after 24 days of incubation. An equal volume of autoclaved wastewater was inoculated as a control.

8.4. Isolation and Growth of M1 Utilizing Microorganisms. The batch cultures capable of degrading M1 were used for further enrichment and isolation of M1 utilizers according to a method previously described.¹⁴ The pure isolates were identified by amplifying and sequencing the urea gene sequences using primers 3C, 5-urea-1, 2-urea-2, 2-urea-2, 2-urea-1, and (7.E, 5-urea-2, 2-urea-2, 2-urea-2, 2-urea-2, 2-urea-1). The isolates were grown in M: M containing M1 or other substrates with indicated concentrations, 0–4 mM, at 30 °C. Medium was supplemented with 0.1% urea ammonium sulfate as a nitrogen source or 10 mM glucose as a carbon source if necessary. The population doubling time was estimated as a method previously described.¹⁴ The optimal growth conditions of the M1 utilizer were tested by culturing the strain in M: M containing 1 mM of M1 under various pH, temperature, and salinity conditions. The pH values were adjusted by phosphoric acid or 1 M sodium hydroxide.

8.5. Isotope Labeling Experiment. M1 was dissolved in H₂¹⁴ or H₂ at a concentration of 0.1 M as the stock standard. The M1 utilizer NyZ550 was inoculated into 100 mL of M: M containing 1 mM M1 and grown to the late exponential phase (OD_{500 nm} of approximately 0.05) at 30 °C. Then, cells were harvested by centrifugation (3000×g, 10 min, 4 °C) and washed twice with 50 mM Tris-H2I buffer, pH 4.0. The cells were divided into two equal volumes, one of which was incubated with 500 μM M1 dissolved in H₂¹⁴ and the other with 500 μM M1 dissolved in H₂. The reaction mixtures were incubated at 30 °C for 2 h and the products were analyzed by ¹⁴C-MSCM. All reactions were performed in triplicate.

8.3. Enzyme Assays. Strain NyZ550 was grown on M1 and harvested as described above. The pellet was suspended in M: M and then lysed by sonication. The resulting cell lysate was centrifuged at 5,000×g for 10 min to remove debris, and the supernatant was used for enzyme assays. The reaction mixtures (0.5 mL containing 500 μg of protein from the crude extract and 0.5–1 mM M1 in 50 mM Tris-H2I buffer, pH 4.0) were incubated at 30 °C. Samples were removed at appropriate intervals and analyzed by ¹⁴C-MSCM, gas chromatography–mass spectrometry (GC–MS), and ion chromatography.

The spectrum change during the reaction was recorded using a 840 nm spectrophotometer, Shimadzu UV-2602S, Norwalk, CT.

7. Analytical Methods. M1 and G were quantified using an HPLC system, Agilent 1100 Infinity II connected in series with triple quadrupole mass spectrometry, Agilent 6490B, Agilent 5390 system. The HPLC system was equipped with an HPLC column, 5 μ m \times 150 mm \times 4.6 mm. Mobile phase A was ultrapure water containing 10 mM ammonium formate, adjusted to pH 5.5 with formic acid, and mobile phase B was methanol. The HPLC gradient started at 5 min, which was held for 5 min, then increased to 75% B at 4.5 min, where it remained until 7 min, and was then reduced to 5% B at 15 min before a 5 min re-equilibration. The binary mobile phase was set to a flow rate of 0.5 mL/min, and the injection volume was 20 μ L. Data analysis, electrospray ionization was used to generate two transition ions from analytes in the positive ionization mode, one transition ion for quantification and the other for confirmation. The following ion transitions, m/z were used: M1 + 105 and + 103, G + 105 and + 103. The ionization parameters were set as follows: the desolvation gas temperature was 100 °C, the drying gas flow was 4 L/min, the nebulizer pressure was 5 psi, the capillary voltage was 3 kV, and the collision energy was 10 eV. Quantification was achieved using the external standard method and Agilent MassHunter Quantitative Analysis software, version 11.0. The calibration curves were generated by calculating the peak area versus the concentration of each target compound. The limits of detection were 10 and 25 ng/L for M1 and G, respectively.

The Agilent 1100 Infinity II-Agilent 5390B HPLC system was used for qualitative analyses of G and other compounds, as indicated. The HPLC system was operated as described above, and GC-MS was performed using an electrospray source in the positive mode with a mass range of 50–500 m/z. The ionization parameters were set as follows: the desolvation gas temperature was 100 °C, the drying gas flow was 5 L/min, the nebulizer pressure was 5 psi, and the capillary voltage was 3 kV.

GC was analyzed with an Agilent 7890A ion chromatograph, Thermo Fisher Scientific, composed of a dual-piston pulse infusion pump system, eluent generator, and digital conductivity detector. The separation was performed by a Dionex IonPac 250 column, 5 \times 150 mm maintained at 100 °C and a conductivity detector using an external calibration. The eluent solution contained 10 mM methanesulfonic acid at a flow rate of 1 mL/min for 50 min. The injection volume was 20 μ L. In addition, GC was also identified using a GC-MS system under the following conditions: an Agilent 5973 column, 1.0 m \times 0.5 μ m \times 0.5 mm was used and interface and source temperatures were 100 °C. The oven temperature procedure was 50 °C for 5 min, increased to 150 °C at 100 °C/min, held for 5 min, then increased to 300 °C at 100 °C/min.

8. Genomic DNA and Transcriptome Sequencing. The M1-grown NyZ550 cells were collected, as described above, and their genomic DNA was extracted by an Omega Bio-Labs Microbial DNA Isolation Kit, Promega Laboratories, Inc. The total DNA of NyZ550 was sequenced by an Illumina HiSeq 2500 platform with the paired-end mode. The sequenced reads were assembled using Velvet software. The draft genome sequence of strain NyZ550 was deposited in the NCBI database under the project accession number JG944474.

train NyZ550 cells grown on glucose, 89 group- or M1 group- were collected and sent to the PersonalBio Technology Co., Ltd., Shanghai, China for NGS sequencing. Total RNA was extracted using a RNeasy lysis method, RNeasy lysis, resulting in RNA samples with a concentration of more than 50 ng/ μ L. Then, RNA libraries of 100–100 bp were constructed and sequenced by the Illumina platform. The adapter sequences and low-quality reads from the raw data were removed before gene expression analysis. The gene expression levels were evaluated as fragments per kilobase of transcript per million fragments (FPKM). The fold change of gene expression was calculated by comparison of the FPKM values of the M1 group to those of the 89 group. The differentially expressed genes were identified by a log₂ fold change \geq 1, p-value \leq 0.05. The transcriptomic raw data were deposited to the NCBI database with accession numbers GSE1555734 and GSE1555733.

9. Degradation of M1 by a Microbial Culture. The G hydrolase gene, guh, GenBank accession number M145100 was synthesized by PersonalBio Technology Co., Ltd., Shanghai, China. The guh gene was amplified by the primers guh-F, 5'-GCTGGGTTGGGTTGTTGGT-3' and guh-R, 5'-GTTGTTGGTGGTGGTGGT-3' and then inserted into the pEJ(+5) vector by homologous recombination. The guh was expressed under the control of a tac promoter. The construct pEJ(+5-guh) was then introduced into *P. putida* ATCC 49619 by electrotransformation, as described previously.¹⁵ Train ATCC 49619 carrying pEJ(+5-guh) was grown in 50 mL of 8# medium containing 10 μ M tetracycline at 100 °C to an optical density at 500 nm (OD_{500 nm}) of 0.1. *Aminobacter* sp. strain NyZ550 was grown in 50 mL of M1 medium containing 1 mM M1 to the exponential phase. The two cultures were harvested by centrifugation and washed twice with 50 mM Tris-HCl, pH 4.0 and resuspended in M1 to a density of 1×10^4 cells/mL. The ATCC 49619 cell suspension was divided into two parts, with one part was used for biotransformation of G and the other mixed with equal number of NyZ550 cells, approximately 5×10^7 cells in total, which was used as a microbial culture for biotransformation of M1. The biotransformation cultures were incubated in a shaker at 100 rpm, 100 °C. Samples were removed at appropriate intervals and analyzed using HPLC-MS. Control experiments were performed using ATCC 49619 cells carrying the empty pEJ(+5) vector. All the assays were tested in triplicate.

RESULTS AND DISCUSSION

9.1. M1 and G in Samples from Wastewater. The concentrations of M1 and G in wastewater samples were measured by HPLC-MS. G was shown in M1 was detected in all samples with a large variation in concentrations, ranging from 0.1 to 45.5 μ g/L. G, a known metabolite of M1, was detected in 7 of the 10 samples with concentrations ranging from 0.5 to 545.5 μ g/L. The levels of M1 and G in each environmental sample exhibited a negative correlation. In the nine samples with high G concentrations, the concentrations of M1 were merely in the range of 0.1–4 μ g/L, while the remaining three samples, H, N, and Z in which G was not detected had high M1 concentrations (3.1–45.5 μ g/L). This result is consistent with previous observations in samples from H and Z and is likely due to

Table 1. Concentrations of M¹ and M² in the Wastewater of 11 Municipal WWTPs in China^a

sites	concentrations, μgF8 ^{-b}	
	M ¹	M ²
#eiling, #-	0.33 ± 0.05	55.05 ± 53.0+
2hang hou, 2Z-	0.70 ± 0.05	0.5+ ± +.0
>uang hou, >Z-	0.++ ± 0.0+	(75.(+ ± +(5.44
Hang hou, HZ-	0.7 ± 0.0'	44.53 ± +(4.54
8an hou, 8Z-	+.' ± 0.' (547.57 ± .. 7.05
Nanling+, N=+	5.05 ± 0.5	' 73.(± 5+.. 7
Nanling. , N=.	. (3.. ± (.4(N6
:hanghai , :H-	' .3 ± 0.5'	' 3'.7' ± +.. 4'
:hilia huang , :Z-	' .0.5 (± +.. +5	N6
&u!i , &H-	(45.5 ± 33.53	N6
Hiamen , HM-	+.(± 0.0'	545.5 ± +.' 4
Auhang , AH-	+.. (± 0.0(. 44.+ (± 73.' 7

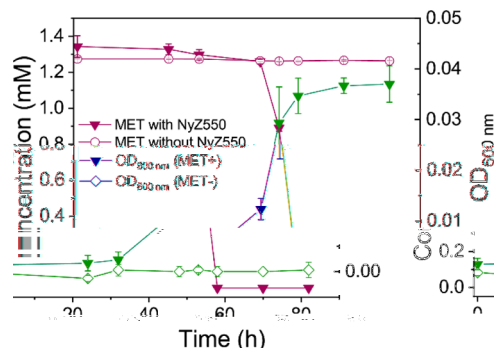
^aN6 not detected. ^bthe data shown are average values of replicate samples, n = 3.

the distinct ability of microorganisms to convert M¹ into M² in wastewater.

Distinct M¹ Degrading Capacity of Indigenous Microorganisms from Wastewater. To compare the M¹-degrading abilities of indigenous microorganisms from different wastewater samples, batch culture biodegradation tests were performed. M: M containing M¹, from 0.7 to +.5 mM was inoculated with wastewater from different sites, and M¹ concentrations were measured over time (Figure 1). Most complete degradation of M¹ was observed in nine cultures within +0 days, whereas three cultures, #eiling, N=., and :Z- showed a poor capacity to degrade M¹ within +. days. Interestingly, these three cultures were derived from the three original samples in which M² was not detected but had high concentrations of M¹, #eiling, N=., and :Z-, as listed in Table 1. The +. cultures were further transferred after +. days to fresh media with +. inoculum and cultured under the same conditions. Similar degradation behaviors were observed in the second transfers as in the first ones, except for sample :Z-, which showed a complete degradation of M¹ but over a longer period, +. days, indicating a possible microbial adaption or evolution for M¹ degradation. Cultures #eiling and N= were still unable to degrade M¹ even after extending the culturing period to .5 days. No evident elimination of M¹ was observed in the controls of the autoclaved wastewater samples, indicating that M¹ was not transformed through an abiotic manner under the conditions used.

M¹ degrading capacity of indigenous bacteria were isolated from wastewater after selective enrichment on M¹ of the cultures exhibiting M¹ degrading capacity, five phylogenetically related *Aminobacter* sp. strains with the ability to grow on M¹ were obtained from different samples, and their 16S rDNA gene sequences were deposited in GenBank. These strains exhibited a similar degradation behavior for M¹, and therefore, an isolate from sample #eiling, designated *Aminobacter* sp. strain NyZ550- was used for further analyses.

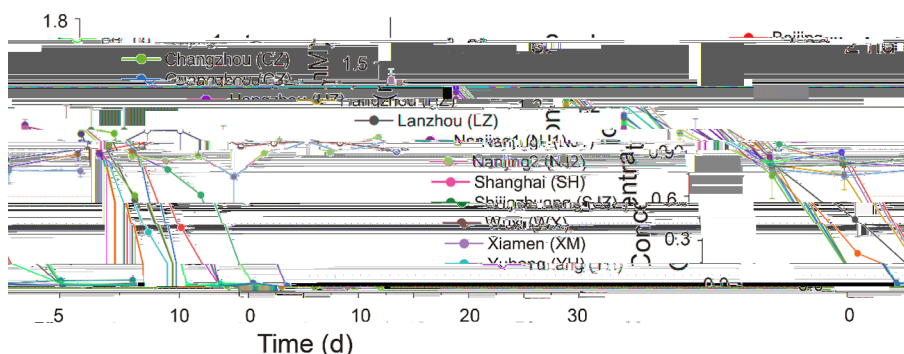
Strain NyZ550 was shown to almost completely degrade M¹ within 50 h with an increase in cell biomass (Figure 2). No degradation of M¹ was observed in



growth of *Aminobacter* sp. strain NyZ550 on M¹ as a sole source of carbon, nitrogen, and energy. Controls were conducted without inoculation or without addition of M¹. Growth is indicated by the increased optical density at 500 nm (OD_{500nm}), and the concentrations of M¹ are shown in fuchsia lines.

the control without inoculation of strain NyZ550, and no growth was observed in the control without the addition of M¹. The results clearly established that the M¹ loss was due to biodegradation by strain NyZ550. The strain grew well at temperatures of +5–'0 °C, preferred neutral or alkaline conditions, pH 5.0–+0.0-, and tolerated salinity up to +. as NaCl, w/v <-. Figure 3.

Strain NyZ550 grows on a variety of Nitrogenous Substrates. Nitrogen availability is often a growth-limiting factor for bacteria, and strains of *Aminobacter* are known for their versatile abilities to utilize nitrogenous compounds, thereby conferring them a selective advantage in the environment.^{3, 4} Here, we tested the growth of strain NyZ550 on various nitrogenous compounds, mM-analogous to M¹ or derived from M¹. The results showed



M¹ degrading capacity of the microbial consortia from wastewater of 11 sites in China. The first transfer, first- and second transfer were performed in minimal salt media containing M¹.

that strain NyZ550 was able to use trimethylamine, 6M?, methylamine, and +-methylbiguanide as sole carbon, nitrogen, and energy sources for its growth, Figures 1 and 2.

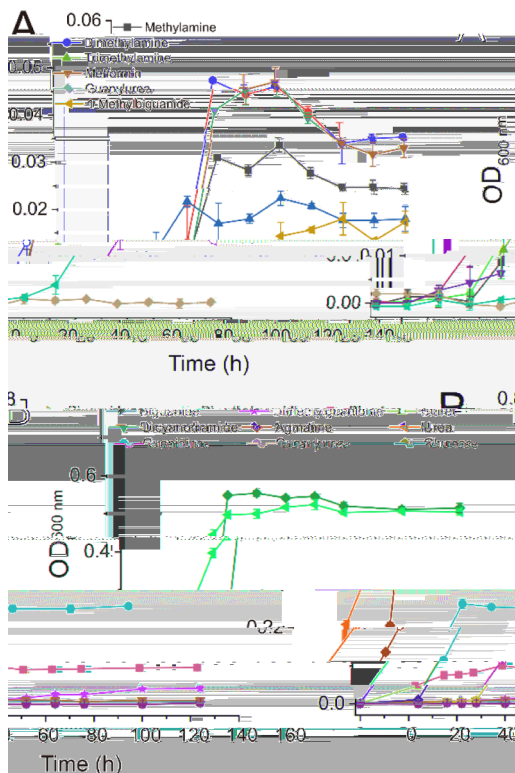


Figure 1. Growth of *Aminobacter* sp. strain NyZ550 on a variety of nitrogenous compounds. The structures of these compounds can be found in Figure S1. 2-compounds, 1 mM each- indicated were used as the sole source of carbon and nitrogen. 3-compounds, 1 mM each- indicated were used as the sole source of nitrogen, and 10 mM glucose was used as the carbon source.

doubling time of strain NyZ550 grown on these substrates was 0.8 ± 0.1 , (4 ± 0.1) , 5.5 ± 0.5 , and 4.5 ± 0.5 h, respectively. The low biomass yields could be due to the inefficiency in transportation, un-optimized catabolic pathway, or toxicity caused by the substrates. Other nitrogenous compounds, including guanidine, biuret, urea, and agmatine, could only be used as the sole nitrogen source for growth by strain NyZ550, Figure

3. However, no clear growth was observed using biguanide, dicyandiamide, or the M1" degradation product 9? under the same culture conditions.

5. Identification of the Catabolic Product 9?. Considering the extremely high proportion of 9? versus M1" in the sample 1H, from which strain NyZ550 was isolated, it was assumed that 9? was a degradation metabolite. Therefore, identification of metabolites produced during growth of strain NyZ550 on M1" was performed using 982-G) C M. 9? product peak with a retention time of 4.7 min was confirmed to be 9? based on comparison of the retention time and mass spectra, 0M M HP^M+0'.05+5, mass accuracy +7 (parts per million ppm) error- with those of authentic 9?, Figure 4. During the growth of strain NyZ550, a stoichiometric accumulation of 9? was observed in the culture with the disappearance of M1", but 9? was not further degraded, even after culturing for an extended period after M1" was transformed, Figure 5. Moreover, strain NyZ550 was unable to grow on 9? with or without the addition of glucose, indicating that 9? could not be used as the source of carbon or nitrogen for growth, Figure 6. These observations established that 9? was a dead-end product in M1" degradation by strain NyZ550. This may well explain the accumulation of 9? in the wastewater, sample 1H- from which the strain NyZ550 was isolated. This same catabolic process is likely to be present in other samples with high 9? concentrations in this study.

3. Incorporation of the ¹⁵N Atom of 9? into 9? during M1" Degradation. To determine the source of the oxygen atom incorporated into 9? and the reaction type for the formation of 9? from M1" degradation, biotransformation assay was performed to convert M1" into 9? by strain NyZ550 in H₂O, and the product was analyzed by mass spectrometry. The molecular ion of ¹⁵N atom-labeled 9? (m/z +05) appeared at a position two mass numbers higher than the corresponding ion of unlabeled 9? (m/z +0), Figure 5. The observed product ratio of ¹⁵N labeled and unlabeled 9? was about 0.05, and this is likely because of the incorporation of H₂O within the cells. The result indicated that the ¹⁵N atom of H₂O was incorporated into the 9?, thus clarifying that the reaction proceeded via hydrolysis.

7. Identification of DMA as Another Initial Degradation Product. The evidence above dictates the biodegradation of M1" by NyZ550 with the accumulation of

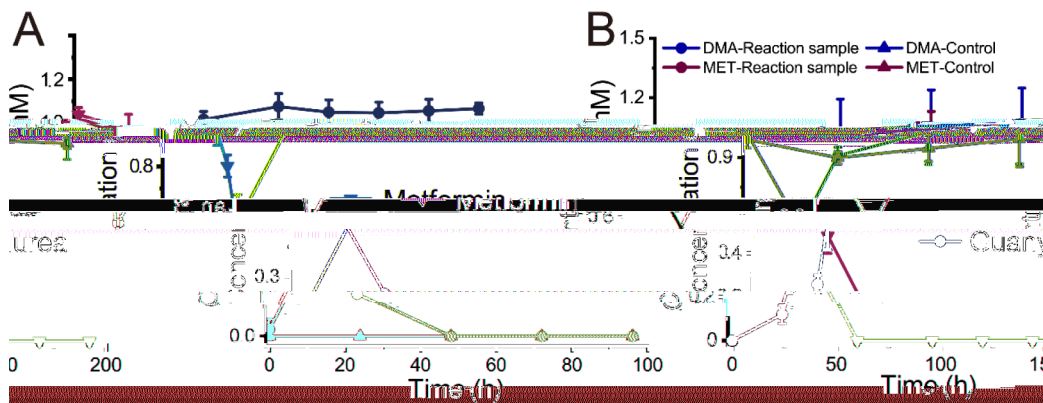


Figure 2. Identification of the M1" degradation products 9? and 6M?. 9? Quantification of 9? formation along with the degradation of M1" during the growth of strain NyZ550. 6M? Quantification of 6M? formation along with the degradation of M1" in crude enzyme assays. The reaction mixture contained 10 mM H₂O buffer, pH 4.0, 1 mM M1", and 0.5 mg of crude enzyme prepared from M1"-grown NyZ550 cells.

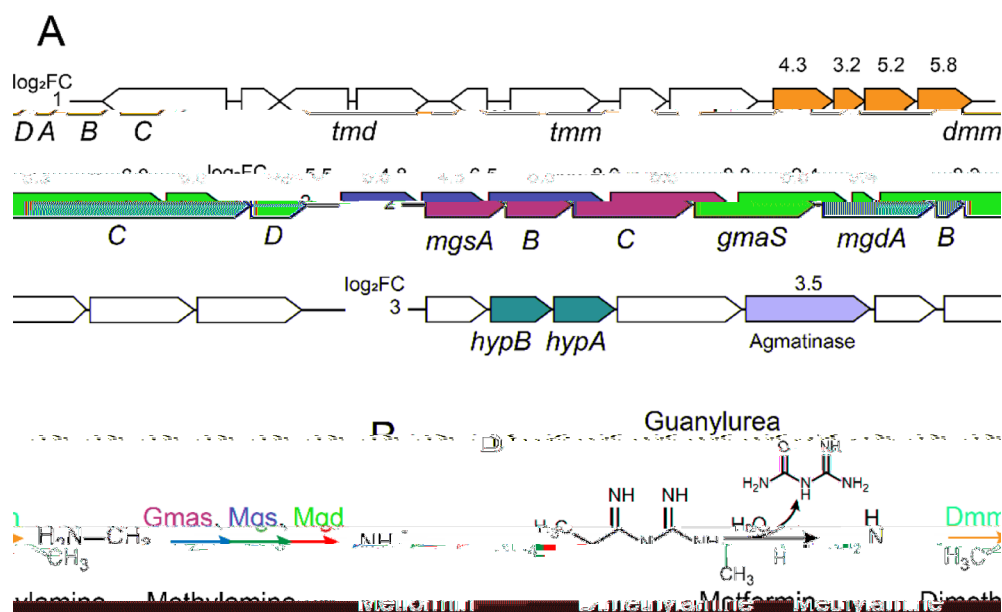


Figure 5. Proposed M1 degradation pathway and involved genes in *Aminobacter* sp. strain NyZ550. (A) Gene clusters involved in catabolism of M1. The transcriptional levels of genes in the M1 degradation pathway are upregulated, as indicated by the log₂ fold change values upon each gene. C2, fold change. 2 clusters + and - encompass the methylamine catabolic genes. - tmm, trimethylamine monoamine oxidase; - dm, trimethylamine N-oxide demethylase; - DmmABC, 6M monamine oxidase; - gmas, -glutamylmethylamide synthetase; - Mgs#2, N-methylglutamate synthase; and - Mgd#26, N-methylglutamate dehydrogenase. 2 cluster + contains the genes encoding the nicotinic incorporation proteins, - Hyp#- and agmatinase. (B) Proposed M1 degradation pathway in strain NyZ550.

the dead-end product 6M^+ , but the exact intermediates for supporting the growth of NyZ550 are unknown. Two possible pathways for bacterial transformation of M1⁺ to 6M^+ were proposed (Figure 5). One was initiated by demethylation, with the intermediates including α -methylbiguanide and biguanide. Although α -methylbiguanide was previously detected in batch experiments with activated sludge from W&S,⁷ in this study, it was not observed in growth or biotransformation assays when compared to standard in 82-G) C M: analysis. Another possible pathway was the direct hydrolysis of M1⁺ with the probable release of 6M⁺. Therefore, we performed an enzymatic assay to detect 6M⁺ using the crude enzyme prepared from M1⁺-grown NyZ550 cells. The activity of the crude enzyme on M1⁺ was confirmed by the decrease in the characteristic absorption of M1⁺ at 295 nm during the enzymatic reaction (Figure 3), as well as the formation of product 6M^+ in the reaction mixture (Figure 3, 2, 6-). Most importantly, a significant amount of 6M⁺ was detected in the reaction mixtures by ion chromatography (Figure 4- and 2-M: (Figure 7-). Consumption of 0.0 ± 0.0 mM M1⁺ led to the formation of 0.0 ± 0.4 mM 6M⁺ in the reaction mixture within 3 h (Figure 7-). 6M⁺ was a favorable growth substrate for strain NyZ550 but 6M^+ was not (Figure 7-). Furthermore, we observed that an equimolar amount of 6M⁺ and M1⁺ resulted in an equal biomass accumulation of strain NyZ550 (Figure 4-), indicating that 6M⁺ was the only intermediate supporting bacterial growth.

Genes Involved in M⁺T Degradation in Strain NyZ550. To further elucidate the degradation mechanism of M1⁺ by strain NyZ550, complete genome and comparative transcriptome analyses were carried out to identify the genes in response to M1⁺. A total of 544 repressed genes were detected, and of them, 103 genes were upregulated, and 10 genes were downregulated (Figure 6). KEGG analysis showed that the upregulated genes were significantly enriched

in pathways of photosynthesis, flagellar assembly, ribosome, glycolate and dicarboxylate metabolism, methane metabolism, one-carbon pool by folate, and glycine, serine, and threonine metabolism (Figure 6). In addition, KEGG annotation analysis also showed that the upregulated genes were enriched in the tricarboxylic acid cycle and one-carbon metabolism, implying that these processes were likely to participate in M1⁺ degradation.

3.8.1. Genes Involved in the MET Degradation Pathway of Strain NyZ550. Based on the genomic annotation, there were two gene clusters in the genome of strain NyZ550 encoding the methylamines degradation (Figure 5). One of the clusters encoded the enzymes for catabolism of trimethylamine (tmm, trimethylamine monoamine oxidase and dm, trimethylamine N-oxide demethylase- and 6M⁺ (Dmm#2, 6M⁺ monoamine oxidase-), while another cluster encoded the enzymes for catabolism of methylamine (α -glutamylmethylamide synthetase Mgs#2, N-methylglutamate synthase and Mgd#26, N-methylglutamate dehydrogenase-. The transcriptome data showed that the transcription of dmm, gmas, mgs, and mgd genes was highly induced during the growth of strain NyZ550 on M1⁺ (Figure 6), indicating that they encoded the enzymes responsible for the catabolism of 6M⁺, a catabolic intermediate of M1⁺. Moreover, assimilation of 6M⁺ in strain NyZ550 was most likely coupled to the serine cycle (Figure 6)- due to the significant induction of its pathway genes and formate-tetrahydrofolate catabolic genes (Figure 6)-.

In addition, the gene encoding the enzyme for hydrolysis of M1⁺ has not been identified. We analyzed the proteins encoded by the 103 upregulated genes in the KEGG protein families and domains database to screen potential M1⁺ hydrolases. It turned out that a putative agmatinase was attractive due to its potential role in hydrolysis of the C-N bond. Agmatinase catalyzes the cleavage of the C-N bond of

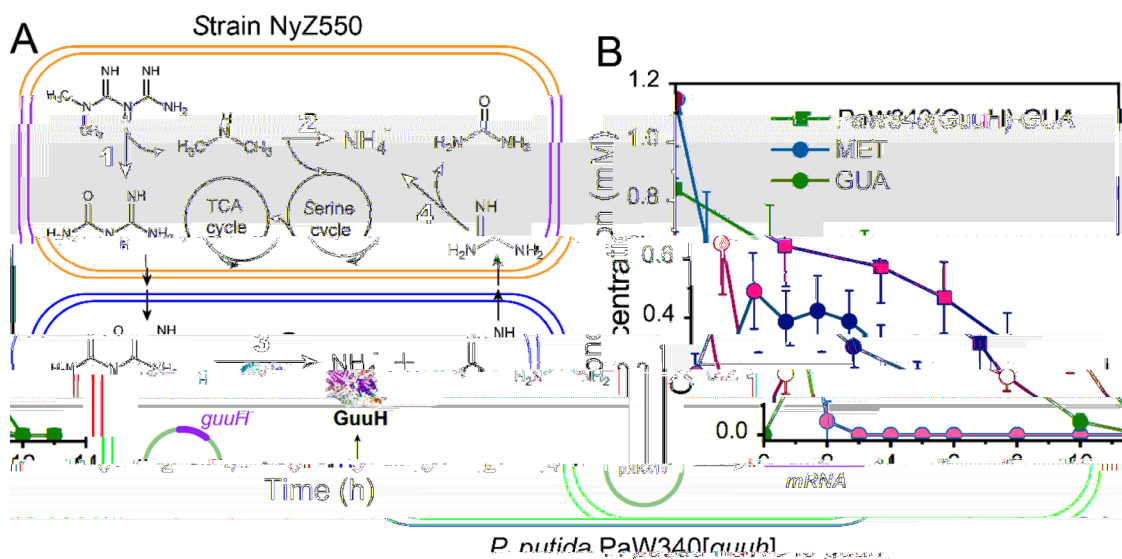


Figure 5. (A) Schematic diagram for the roles of the bacterial strains in the mixture. The key processes in M1 and >9? degradation are shown: (1) hydrolysis of M1, (2) hydrolysis of >9?, (3) hydrolysis of guanidine, (4) degradation of M1 and >9? by the mixture. (B) Concentration (mM) of MET, GUA, and PaW340(GuuH) GUA over time (h). The inset shows mRNA levels for *guuH* and *guuF*.

agmatine with the formation of putrescine and urea. In this reaction, an oxygen atom from H₂O is incorporated into urea, which is similar to the reaction mechanism for hydrolysis of M1. Arginases are reported to be nickel or manganese-dependent hydrolase. Two genes encoding the nickel incorporation proteins, *HypA* and *HypB*, which is required for the loading of divalent metal ion on arginase are adjacent to the upregulated arginase gene in strain NyZ550, Figure 5. Further investigations of M1 hydrolase in biochemistry, catalytic mechanism, evolution, and ecological distribution would be helpful for a better understanding of the environmental fate of M1. Based on the metabolite analyses and results from transcriptomic sequencing, the degradation mechanism of M1 and its intermediates by strain NyZ550 was inferred, as shown in Figure 5.

>9? hydrolase was absent from the NyZ550 genome, which is consistent with the accumulation of >9? during degradation of M1. Although >9? was not degraded by strain NyZ550, guanidine, a known catabolic intermediate of >9?, was readily utilized as a nitrogen source by strain NyZ550, Figure 5. Putative arginase of strain NyZ550 was homologous to the guanidinase *gdmH*, 54% identity, from cyanobacterium *Synechocystis* sp. PCC 5407. It likely catalyzed the conversion of guanidine into ammonium and urea.

3.8. Other Important Biological Processes Related to MET Degradation. M1 is a hydrophilic compound and its degradation by strain NyZ550 is dependent on intracellular enzymes, and therefore, M1 must be transported into the cell before it can be degraded. This process mainly involves energy metabolism, signal sensing, binding, and transmembrane transportation. As shown in Figure 6, some bacterial chemotaxis genes were upregulated during growth on M1, which is beneficial for accessing and transporting substrates. Genes encoding 2 transporters and substrate-binding proteins were also upregulated, suggesting that they are likely involved in the transmembrane transportation of M1. Additionally, an upregulated gene encoding the putative

guanidinium exporter is likely contributed to minimizing the stress exerted by endogenous M1 or its catabolic products.

3.9. Degradation of M1 and >9? by a Constructed Bacterial Mixture. Considering the risk of frequently detected >9? accumulation in the environment, the complete degradation of >9? and the accumulation of the dead-end product >9? in the degradation of M1 by strain NyZ550 in this study, >9? degradation is also necessary for bioremediation of M1-contaminated sites. The results here regarding M1 catabolism and the genes involved in strain NyZ550 clearly indicated that the conversion of >9? was the catabolic bottleneck. Thus, a bacterial mixture was constructed for complete degradation of M1 and >9?, consisting of strain NyZ550 and *P. putida* PaW340 (harboring a >9? hydrolase gene, *guuH*, Figure 5). *guuH* converted >9? into guanidine, which was subsequently utilized as a nitrogen source by strain NyZ550, Figure 5. *P. putida* PaW340 cells expressing >9? hydrolase, designated *P. putida* PaW340-*guuH*, degraded >9? at a rate of 4.5 nmol mg protein⁻¹ h⁻¹, Figure 5. The bacterial mixture composed of strain NyZ550 and *P. putida* PaW340-*guuH* exhibited the capacity to eliminate both M1 and >9?. M1, 100 mM, disappeared within 1 h, accompanied by the significant accumulation of >9? with a maximum concentration of 0.5 mM at 1 h. Complete degradation of >9? was observed after 24 h, Figure 5.

The constructed bacterial mixture capable of complete degradation of M1 and >9? in water provides a potential strategy for bioremediation of M1-contaminated environments. The degradation of >9? appears to be the rate-limiting step for M1 degradation in the environment, due to the fact that it is often detected as a dead-end product of M1. The results presented here suggest that the introduction of a >9? degrader can be a potential solution for accelerating complete removal of M1 contamination through the combined effect of indigenous M1 degraders in ecosystems, such as strain NyZ550.

3.10. Implications for M1-Contaminated Environments. Environmental exposure to M1 is increasingly

recognized as an emerging issue threatening human health and ecological safety.^{3,4,5,5} Clearly, microorganisms are the vital drivers of M1 degradation in the environment. Here, based on an investigation of M1 contamination in wastewater from 8& and the M1 degradation capacity of microorganisms from these wastewater samples, we isolated and characterized a M1-degrading bacterium *Aminobacter* sp. strain NyZ550. Currently, the primary biodegradation pathways for M1 are predicted based on identifying the M1 metabolites in situ or derived from degradation by microbial consortia.^{7,10} One of the possible degradation pathways involves the sequential removal of two urea, giving the first intermediate dimethylguanidine and subsequently 6M, Figure 5. Another conceivable pathway results in the production of biguanide through double deacylation and further formation of 9 as a dead-end product⁷, Figure 5. Based on the results in this study and comparison with previously predicted pathways, a novel pathway for partial catabolism of M1 in strain NyZ550 was proposed, Figure 5. In this pathway, M1 was directly hydrolyzed to produce 9 and 6M. Although 9 was not catabolized further, 6M was assimilated through a complete set of enzymes for methylamine degradation in strain NyZ550, Figure 5, which is echoed by the fact that strain NyZ550 was able to grow on 6M, as shown in Figure 7. The results support the observations of frequent accumulation of 9 in the environment¹⁰ and suggest a possible role of this pathway in environmental transformation of M1. In the future, functional characterization of the catabolic enzymes involved in M1 degradation at the genetic, biochemical, and ecological levels would broaden our understanding of the environmental fate of the widely used antidiabetic drug M1.

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* Supporting Information

The Supporting Information is available free of charge at <https://pubs.acs.org/doi/10.1021/acs.est.1c03557>.

Details of growth, biotransformation product identification, and transcriptome, Figure 6.

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Notes

The authors declare no competing financial interest.

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