

# Environmental Fate and Toxicology of Clomazone

April Van Scoy and Ronald S. Tjeerdema

Department of Environmental Toxicology, College of Agricultural & Environmental Sciences, University of California, Davis, CA 95616-8588, USA

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Corresponding Author

April R. Van Scoy

Department of Environmental Toxicology

University of California, Davis

One Shields Ave

Davis, CA 95616

Phone: (530) 752-2534

## 1 Introduction

The herbicide clomazone (2-(2-chlorophenyl)methyl-4,4-dimethyl-3-isoxazolidinone; CAS 81777-89-1; Fig. 1) was first approved for use in 1986 (US EPA 2007). It is produced by the FMC Corporation under the trade names that include Command® and Cerano® 5 MEG (Tenbrook et al. 2006).

Clomazone is the only isoxazolanone herbicide registered for use within the United States (US EPA 2007). It is used for annual control of broad-leaf and grassy weeds such as barnyard grass (*Echinochloa crus-galli*), crab grass (*Digitaria spp.*), foxtails (*Setaria spp.*), and others that infest soybean, tobacco, rice and other row crops (Scott et al. 1995; Lee et al. 2004; Schocken 1997). Within the USA, approx. 503,487 kg of active ingredient is applied each year (US EPA 2007). It is formulated as an emulsifiable concentrate and microencapsulated flowable granule (5% clomazone) and is applied either pre- or post-emergence (CDPR 2003; US EPA 2007).

Clomazone is highly water soluble and weakly to moderately persistent in soils with half-lives ( $t_{1/2s}$ ) ranging from 5 to 60 days. Because of its water solubility, the potential impact of clomazone on surface water, groundwater and aquatic organisms is of great concern. In this paper, we have reviewed the relevant literature and address clomazone's chemistry, environmental fate and toxicity.

## 2 Chemistry and Physicochemical Properties

Clomazone is an isoxazolanone herbicide containing a chloroaromatic ring (Fig. 1). When pure, clomazone is a crystalline solid (CDPR 2003). At room temperature it is highly soluble in water

and has a low-to-moderate affinity for soil. This herbicide is denser than water, and is susceptible to microbial degradation. The physiochemical properties of clomazone are presented in Table 1.

### **3 Environmental Chemodynamics**

#### ***3.1 Soil***

Clomazone is not expected to bind to soils strongly given its relatively low  $K_d$  and its hydrophilic nature. However, sorption to various soil types (with varying temperature and moisture) has been investigated. Mervosh et al. (1995b) observed that a concentration of ca. 9 mg/kg of  $^{14}\text{C}$ -clomazone sorbed to a silty clay loam soil; such sorption was independent of temperature, and soil moisture content had a minor sorbtive effect. Although overall soil sorption is low, the agent has a higher affinity for binding to humic acid than to whole soil (Gunasekara et al. 2009). Furthermore, it appears that the presence of black carbon or burned residues, in fire-affected locations, increase the sorption of this herbicide (Gunasekara et al. 2009). Loux et al. (1989) determined that adsorption was dictated by organic matter rather than by clay content;  $K_d$  values for clomazone ranged from 0.47 for silt loam to 5.3 for loamy sand.

Half-lives and desorption coefficients were determined for clomazone in four Tasmanian soils. A first-order half-life ( $t_{1/2}$ ) for ferrosol (clay loam), kurosol (loamy sand), sodosol (silt loam), and vertosol (light clay) soils ranged from 79 to 124 d, respectively. Half-lives derived from the Hoerl equation ranged from 6 to 59 d, respectively; this equation provided a good fit to the measured concentrations (Cumming et al. 2002). Desorption also varied with soil type;  $K_d$  values ranged from 1.7 to 3.6, respectively.

The persistence of clomazone was examined under both conventional and no-tillage practices. Following an initial application rate of 1.4 kg/ha, measurable amounts of clomazone were detected at a soil depth of 0-10 cm, 120-days later (Mills et al. 1989). Soil concentrations of  $124 \pm 54$  and  $30 \pm 12$  ng/g, respectively, were measured following conventional and no-till practices (Mills et al. 1989).

Quayle et al. (2006) applied clomazone to simulated flooded rice plots and measured resulting soil concentrations. Analytical results varied 7.5-fold between the 4 and 48 day post-application samplings. In addition, a measured  $t_{1/2}$  of 14.6 days was attributed to anaerobic conditions. Half-lives of 32.9 and 37.4 d, respectively, in Montana loam and silty clay loam soils were noted by Gallandt et al. (1989). Field half-lives for Tennessee clay loam and loam soils ranged from 5-to-29 d and a  $t_{1/2}$  of 34 d resulted under laboratory conditions (Kirksey et al. 1996); this indicates that environmental conditions affect clomazone's dissipation rate.

### **3.2 Water**

Due to its high water solubility (1,102 mg/L) and relatively low  $K_{ow}$  value, clomazone is expected to concentrate within the aqueous phase; thus, concerns exist for potential impacts on drinking water systems. To investigate risks posed to drinking water, Byers et al. (1995) measured clomazone concentrations in vadose zone waters at depths of 0.3, 0.6 and 1.5 m using tension lysimeters. They found concentrations to decrease (ca. 3-fold) as soil depth increased. In addition, soil treatments no mulch was used, or plastic mulch was used had measurable clomazone concentrations respectively of 0.09 and 0.04 ppb (Byers et al. 1995).

Since flooded fields discharge excess water into surrounding creeks and rivers, there is potential for applied residual organics to contaminate surrounding water bodies. The dissipation of clomazone from floodwaters was studied by Quayle et al. (2006). When Quayle et al. (2006) applied clomazone to small replicated rice plots at a rate of 0.5 L/ ha (i.e., as commercially formulated Magister® containing 480 g/L a.i.), an initial measured mean water concentration of 202 ug/L was produced. However, within 4 d the concentration had decreased to 83 ug/L, and by 19 d the concentration declined to 3 ug/L (Quayle et al. 2006). The  $t_{1/2}$  for chlomazone in this study was 7.2 d. Furthermore, the releasing waters contained 3 ug/L clomazone, which was assessed as having a low toxicity hazard. Two Brazilian rivers, the Vacacaí-Mirim River and the Vacacaí River, were monitored for residues of clomazone, particularly sourced from rice field irrigation. An average level of 4.5 ug/L was measured within 41% of collected samples from the Vacacaí River whereas the Vacacaí-Mirim River had measurable concentrations of 3.7 ug/L in 33% of samples (Marchesan et al. 2007). The higher rate and level of detections in the Vacacaí River were attributed to its larger surrounding drainage area and plot acreage. Zanella et al. (2002) reported residual clomazone concentrations in samples collected from experimental rice fields in the central region of the Rio Grande do Sul, Brazil. During both December 1999 and 2000, samples collected 130 days post application were found to contain clomazone concentrations of 0.9 and 0.2 ug/L, respectively (Zanella et al. 2002).

### ***3.3 Air and Volatilization***

The volatility of various formulations of clomazone from Flanagan silt loam was studied under both moist soil and simulated rainfall conditions. Mervosh et al. (1995c) reported that each of the granular formulations reduced volatilization; small granules (20 to 30 mesh) produced greater volatilization than did those of 14 to 20 mesh. In addition, they found that soil-water content

greatly affected volatilization flux; highly saturated soil resulted in increased flux rates. Compared to others, starch-based formulations reduced off-site movement (Mervosh et al. 1995c).

Thelen et al. (1988) observed volatilization up to 2 weeks post-application in both surface applied or soil-incorporated treatments; surface application resulted in higher volatilization. In addition, the presence of rainfall increased clomazone's overall tendency to volatilize. The off-site movement of vapors from extremely wet soil was observed by Halstead and Harvey (1988). Such vapors traveled as far as 32 m from the application site as measured by phytotoxicity to sunflower and wheat chlorosis. The application rate was a major factor in producing phytotoxic effects at this distance; however soil moisture and wind speed may have contributed to clomazone's transport (Halstead and Harvey 1988). Mervosh et al. (1995a) observed increased volatilization from increasing temperature, but not from soil moisture. Schummer et al. (2010) determined that air samples, collected from a farming site in Northeastern France, contained gas-phase concentrations of clomazone ranging from 0.14 to 0.68 ng/m<sup>3</sup>.

## **4 Environmental Degradation**

### ***4.1 Abiotic Processes***

In buffered solutions (pH 4.65, 7.0 and 9.25, 25°C, 41d) clomazone was found to be hydrolytically stable over the entire test period (Dziedzic 1982). Breakdown of the herbicide was <10% of the initial concentration at each pH, but the natures of the resulting products were not determined. CDPR (2003) reported similar observations, discovering that clomazone was stable

under various pH conditions, as measured after 34-40 d. We conclude that hydrolysis is unlikely to be a major degradation route for clomazone.

Experimental studies, in which the direct and indirect photolytic degradation of clomazone was assessed under California rice field conditions, were conducted by Tomco and Tjeerdema (2012). Clomazone was found to degrade slowly when exposed for 35 d to either natural or artificial sunlight (8W UV lights, exhibited  $\lambda_{\text{max}} = 300 \text{ nm} \pm 50 \text{ nm}$ , 30°C). Half-lives were determined to be 145 and 158 d, respectively, for artificial and natural light exposures (Tomco and Tjeerdema 2012).

Photolytic degradation from surface waters contributes to the dissipation of many pesticides and other xenobiotics. Zanella et al. (2008) investigated the photodegradation rate of clomazone in both distilled and agricultural field water irradiated for up to 120 min; after 60 min, a 6.5-fold higher concentration remained in agricultural than in distilled water. However, clomazone's rate of degradation in agricultural field water was affected by pH (pH 3 caused more efficient degradation than pH 6); the difference between the two pH values was attributed to the photo-Fenton process (Zanella et al. 2008).

It is thought that humic substances act as photosensitizing agents that increase the likelihood of oxidative degradation (Gara et al. 2009). Since clomazone strongly sorbs to organic matter, photolysis would be more dominant under conditions of high humic acid content. To examine this theory, Gara et al. (2009) irradiated ( $\lambda > 300 \text{ nm}$ ) air-saturated Aldrich humic acid (20 ppm) and observed a degradation enhancement and depletion in total organic carbon (TOC);

byproducts included 2-chlorobenzyl alcohol and 2-chlorobenzaldehyde. In addition, irradiated ( $\lambda > 200$  nm, 125 W mercury lamp, 20 min, 27°C) aqueous peroxydisulfate ( $S_2O_8^{2-}$ ) solutions degraded clomazone by >90%. A proposed degradation pathway for clomazone is shown in Fig. 2. Photochemical experiments (aqueous, 25°C, pH 4-5) conducted by Kirmser et al. (2010) used  $SO_4^-$  radicals to degrade clomazone; two products resulted – 2-chlorobenzyl alcohol and 2-chlorobenzaldehyde.

#### **4.2 Biotic Processes**

Biotransformation of clomazone was studied by Helbling et al. (2010). Sludge collected from a pilot-scale wastewater treatment plant was spiked with pesticides and pharmaceuticals; clomazone was spiked at a concentration of 100 ug/L. Both mono- and di-hydroxylated clomazone transformation products were identified (Fig. 3); however the positioning of the hydroxy groups were not confirmed.

Liu et al. (1996) exposed both *Aspergillus niger* (UI-X172) and *Cunninghamella echinulata* (NRRL-3655), a common soil fungus and bacterium, respectively, to clomazone; 95% of the agent was metabolized by *A. niger* (X172). Transformation processes included both aromatic ring hydroxylation and benzylic hydroxylation – with subsequent dehydrogenation, and identified metabolites included 5-hydroxyclozoxazole, hydroxymethylclomazone, 2-chlorobenzyl alcohol and 3'-hydroxyclozoxazole (Fig 4; Liu et al. 1996).

Mervosh et al. (1995a) investigated both the mineralization and microbial degradation of  $^{14}C$ -clomazone in Flanagan silt clay loam soil and found mineralization to be dependent on microbial



activity; mineralization was more active at lower temperatures. According to Mills et al. (1989), microbial degradation of clomazone is favored under neutral soil pH conditions, whereas microbial populations tend to be more abundant under no-till conditions.

In several studies, clomazone has been observed to degrade more rapidly under flooded conditions, suggesting that anaerobic bacteria play an important role in degrading it. One such study was designed to simulate aerobic and anaerobic California rice field conditions and to measure the anaerobic degradation rate for clomazone. Results were that clomazone was anaerobically degraded to produce metabolites within 3 days of its application. In contrast, under aerobic conditions, clomazone residues became soil-sorbed residues (Tomco et al. 2010). Anaerobic and aerobic half-lives of clomazone were reported to be 7.9 d and 47.3 d, respectively. However,  $^{13}\text{C}$ -labeled clomazone was found to significantly mineralize under aerobic conditions (18.6% of applied compound), whereas only 3% was mineralized under anaerobic conditions (Tomco et al. 2010).

TenBrook et al. (2006) suggested that microbial degradation of clomazone could be subject to photolytic enhancement; however, to date this phenomenon has not been experimentally demonstrated. Lack of photolytic assistance was further confirmed by Tomco and Tjeerdema (2012). They found soil microbial degradation to be more relevant than photolysis, and thus, it appears to be the major degradative pathway for clomazone (Tomco and Tjeerdema 2012).

## **5 Toxicology**

### ***5.1 Mode of Action in Plants***

Clomazone is designed to target broad-leaf grasses; however, it has been known to cause toxicity in other plants, where it systemically enters through the roots and shoots and translocates via the xylem (US EPA 2007). Studies have shown that this herbicide impairs the formation of photosynthetic pigments, reducing both chlorophyll and carotenoids and bleaching foliar structures (see section 5.2). It is thought that the metabolite 5-ketoclomazone may be responsible for such toxicity (US EPA 2007). The mode of action, particularly on the methylerythritol-4-phosphate (MEP) pathway, resulting from clomazone, 5-hydroxyclozomazone and 5-ketoclozomazone, has been investigated. Ferhatoglu and Barrett (2006) reported that clomazone and 5-hydroxyclozomazone did not inhibit the MEP pathway in spinach, although they are known to cause plant bleaching; however, 5-ketoclozomazone did inhibit this pathway. In addition, the parent and hydroxylated metabolite did not inhibit synthesis of either chloroplastic isoprenoids or deoxyxylulose 5-phosphate (DXP) within this pathway, whereas 5-ketoclozomazone did. Ferhatoglu and Barrett (2006) concluded that subsequent toxicity and plant bleaching results from the ultimate toxicant 5-ketoclozomazone. However, the exact site of action is in question. Suggested sites include: 1) isopentenyl pyrophosphate isomerase, 2) prenyl transferases, and 3) enzymatic phytylation of chlorophyllide (Duke et al. 1985, 1986; Sandmann et al. 1987).

## ***5.2 Plant Effects***

Since clomazone is extensively used to control weeds in rice culture, many researchers have investigated its impact on rice production and plant growth. Bollich et al. (2000) applied various rates of microencapsulated clomazone (0.28 to 2.2 kg a.i./ha) to both water-seeded and drill-seeded fields. Minimal bleaching of rice seedlings occurred at low application rates, although increased application rates increased bleaching. Furthermore, higher rates also reduced seed head emergence and grain yield; drill-seeded production was more impacted than water-seeded

(Bollich et al. 2000). Webster et al. (1999) observed injury to rice 7d after emergence; however less injury was produced by post-emergence treatments.

The spectrum of clomazone's weed control was investigated by Westberg et al. (1989). Pre-emergence application (at 280 g a.i./ha) controlled 90-100% of weeds (e.g., barnyardgrass, foxtail, crabgrass and velvetleaf) and was more effective at equal rates than was pre-plant incorporated application. Weed tolerance to this herbicide was studied by Liebl and Norman (1991). They found soybean seedlings to be 256 times more tolerant to clomazone than velvetleaf, whereas corn and smooth pigweed were 66 and 13 times less tolerant, respectively. In addition, a reduction in shoot fresh weight and leaf chlorophyll was observed only for velvetleaf.

ElNaggar et al. (1992) applied  $^{14}\text{C}$ -clomazone (1.1 and 2.2 kg a.i./ha) to soybean seeds planted in pots and at a depth of between 1.2 and 2.5 cm; plants were harvested after either 30 or 60 d exposure periods. Clomazone metabolites within plant tissues were identified. It appeared that the major degradation processes included dealkylation and monohydroxylation (Fig 5); subsequent conjugation processes resulted in glycoside formation.

TenBrook et al. (2006) identified glucose conjugation as the main route of phase II detoxification. They also identified metabolites in rice (*Oryza sativa*) and early watergrass (*Echinochloa spp.*) to be conjugated glucosides. Although Norman et al. (1990) did not identify specific clomazone metabolites, they did note that up to 5.9% of recovered residues in the seeds of both soybean (*Glycine max*) and cotton (*Gossypium hirsutum*) were  $\beta$ -glucosides.

Kana et al. (2004) investigated the photosynthetic capability of barley seedlings (*Hordeum vulgare*) that were cultivated on filter paper containing either 0.25 mM or 0.5mM clomazone (12 d, continuous light, 10 °C). Adverse effects included a reduction in chlorophyll (a+b) and carotenoid levels. Kana et al. (2004) concluded that the photochemical processes in this species cannot operate fully due to pigment loss brought on by clomazone toxicity. Similarly, Yasuor et al. (2008) found clomazone and 5-ketoclomazone to illicit greater inhibition of chlorophyll a and carotenoids in susceptible populations of late watergrass (*Echinochloa phyllopogon*) than in resistant populations. Further studies on susceptible cotton (*Gossypium hirsutum* L.) showed a slowing of chlorophyllide to chlorophyll conversion and complete inhibition of carotenoid synthesis (Duke et al. 1991). These findings indicate that clomazone inhibits terpenoid synthesis

### **5.3 Aquatic Organisms**

Although clomazone is highly water soluble, its affinity to partition across biological membranes is minimal, as indicated by its low log  $K_{ow}$  value (Table 1). However, silver catfish (*Rhamdia quelen*) fingerlings, exposed to varying concentrations of clomazone, produced a 96-h  $LC_{50}$  value of 7.32 mg/L; thus, clomazone is more highly toxic to this species than to others, like rainbow trout (*Oncorhynchus mykiss*;  $LC_{50}$ = 19 mg/L) and bluegill (*Lepomis macrochirus*;  $LC_{50}$ = 34 mg/L; dos Santos Miron et al. 2004).

Silver catfish exposed for 192 h resulted in brain and muscle AChE (acetylcholinesterase) inhibition of 47% (at 1.0 mg/L clomazone) and 45% (at 0.5 and 1.0 mg/L concentrations), respectively, within 12-h; hepatic vacuolation was also observed (Crestani et al. 2007). Although, biological responses resulted, fish placed into clean water for either 96- or 192-h did recover. Similar results for brain, muscle and heart tissues from piava (*Leporinus obtusidens*) were

reported by dos Santos Miron et al. (2008); however, AChE recovery only occurred in heart. AChE activity was also studied in *Cyprinus carpio* exposed to clomazone for 7, 30 and 90 days under laboratory and field conditions. Although Cattaneo et al. (2012) observed no alteration in AChE activity in field-exposed fish, laboratory-treated fish showed decreased muscle activity after 7 d.

A teleost species (*Leporinus obtusidens*) was exposed to clomazone at a relevant rice field concentration (0.5 mg/L, 30 d), and both tissue AChE and catalase activities were examined. Moraes et al. (2007) reported differential results for clomazone's activity in this species, to wit, a decrease in brain AChE activity, but a significant increase in muscle tissue. Another effect observed was an increase in liver catalase activity. Moraes et al. (2009) also studied the effects of exposed to commercially formulated clomazone in a rice field (0.376 mg/L a.i., 90 d). They reported results similar to those reported in their 2007 study; however liver catalase activity decreased from the latter exposure.

Clomazone's toxicity to aquatic invertebrates has also been investigated. Mysid shrimp (*Americamysis bahia*) are very susceptible, having a 96-h LC<sub>50</sub> value of 556 ug/L (CDPR 2003). The water flea (*D. magna*; 48-h EC<sub>50</sub>= 5,400 ug/L) and Eastern oyster (*Crassostrea virginica*; 96-h EC<sub>50</sub>= 5,300 ug/L; US EPA 2009) were considerably more tolerant to clomazone's toxicity. Although clomazone is designed to target plants, it is also moderately-to-highly toxic to aquatic organisms.

#### **5.4 Non-Target Fauna**

The effects of clomazone on non-target aquatic species were investigated by Burdett et al. (2001). Midge (*Chironomus tepperi*) first instar larvae, exposed at levels up to 0.288 mg/L or to the anticipated field concentration (0.576 mg/L), displayed no significant adverse effects (changes in emergence, development time or wing length) when compared to thiobencarb. Furthermore, clomazone-treated ponds were found to significantly reduce macrophyte biomass and contain higher populations of both ostracods and corixids, when compared to control ponds (Burdett et al. 2001). Perschbacher et al. (2002) exposed pond phytoplankton and zooplankton to clomazone (0.6 kg a.i./ha), in addition to other rice herbicides; no measurable effects were observed.

Early life-stage effects have also been studied. The soybean cyst nematode, *Heterodera glycines* (females, cysts and eggs) was exposed to 50 and 500 ug/mL clomazone (dark, 25°C, 24 d) to assess hatching effects (Wong et al. 1993). Hatching of eggs was not impacted by the herbicide and results did not differ significantly from those of the controls.

Mortality and behavioral responses resulting from clomazone exposures were examined for spiders. Four species (*Dictyna uncinata*, *Pardosa palustris*, *Philodromus cespitum* and *Theridion impressum*) were exposed to fresh (2 h; approx. 0.4g of a.i. in 100 ml of distilled water) and aged (5 to 20 d) herbicide residues (concentrations unknown). For all four species mortality induced by clomazone exposure was minimal (<20%; herbicide activity declined with aging); however, increased movement was observed in *P. palustris* (Pekar and Benes 2008). Pekar (2002) also noted that this herbicide was harmless to *T. impressum*, thus recommending it for plant protection.

According to the California Department of Pesticide Regulation (CDPR 2003), toxicity studies on rats, various birds and honeybees have shown clomazone to be relatively non-toxic to moderately toxic to these species. Generally, study results have shown that clomazone poses little to no risk to non-target species.

Human adverse effects from exposure to clomazone have also been observed. For example, human erythrocytes have been exposed *in vitro* to a range of clomazone concentrations (0, 100, 250 and 500 ug/L) for 1 h to investigate effects on oxidative stress biomarkers and on AChE activity (Santi et al. 2011); both AChE and catalase activities decreased with each concentration, whereas glutathione (GSH) was not affected.

## **6 Summary**

Clomazone, an isoxazolane herbicide, was first registered for use in 1986 for pest grasses and broadleaf weeds. Although the exact mode of action is still unclear, it is well documented that clomazone causes bleaching of foliar structures; the clomazone metabolite 5-ketoclomazone is regarded to cause the bleaching and to be the ultimate plant toxicant. Although clomazone exhibits low mammalian toxicity and is selective towards certain plant species, studies have shown that it does inhibit AChE and catalase activities. In addition, it has been found to be highly toxic to aquatic invertebrates, in particular mysid shrimp.

Clomazone has a low Henry's law constant and moderate vapor pressure, and thus may volatilize from dry soils. Photolysis represents a minor dissipation pathway; however, clomazone can be photolytically degraded under both direct and indirect conditions. Clomazone has high water solubility, and it is often assumed to undergo hydrolysis easily; unfortunately, this is not the case. Clomazone is stable over a wide pH range and does not hydrolyze. Clomazone has a weak to moderate soil adsorption coefficient; therefore, its affinity to sorb to soil is minimal, rendering it a potential threat to groundwater supplies.

Microbial metabolism is the major degradation pathway, resulting in products such as 5-hydroxyclozoxon, hydroxymethylclomazone, 2-chlorobenzyl alcohol and 3'-hydroxyclozoxon. Although clomazone has not been shown to degrade via hydrolysis, it nonetheless represents a potential threat to aquatic organisms. With this in mind, caution should be taken when applying clomazone or when draining fields that have detectable clomazone residues.

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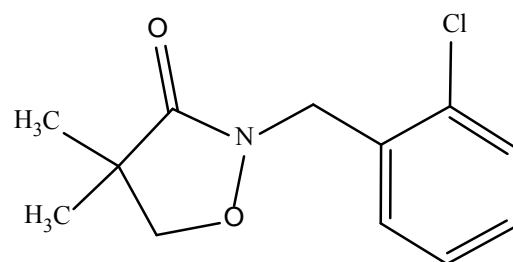


Fig. 1 Clomazone structure

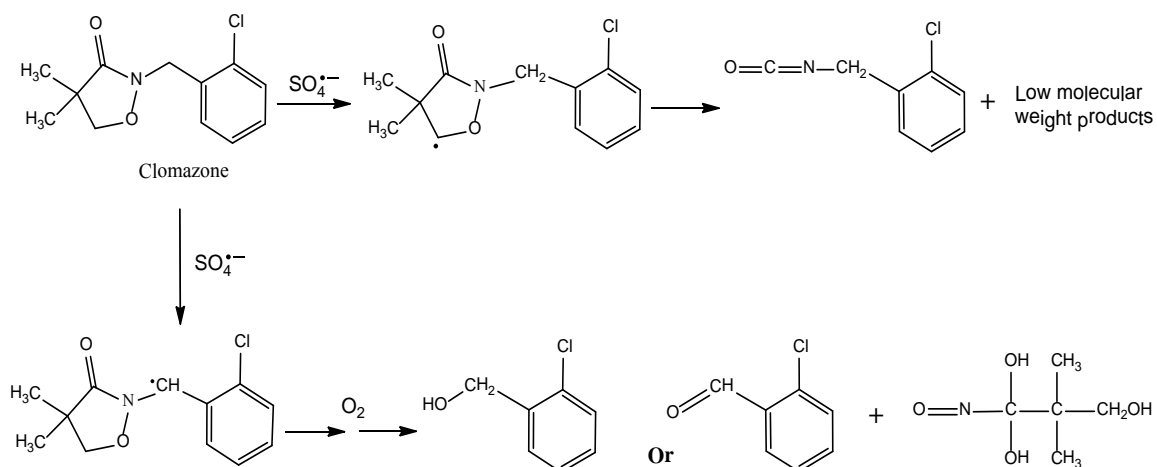


Fig. 2 Photolytic degradation of clomazone with  $\text{S}_2\text{O}_8^{2-}$  at  $\lambda > 200\text{nm}$ , as proposed by David Gara et al. (2009)

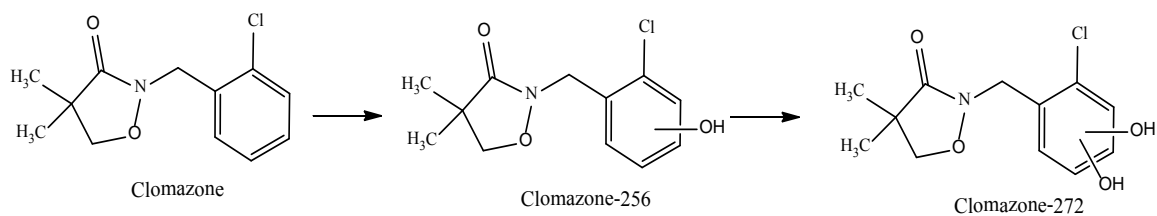


Fig. 3 Suggested microbial transformation pathway for clomazone (adapted from Helbling et al. 2010)





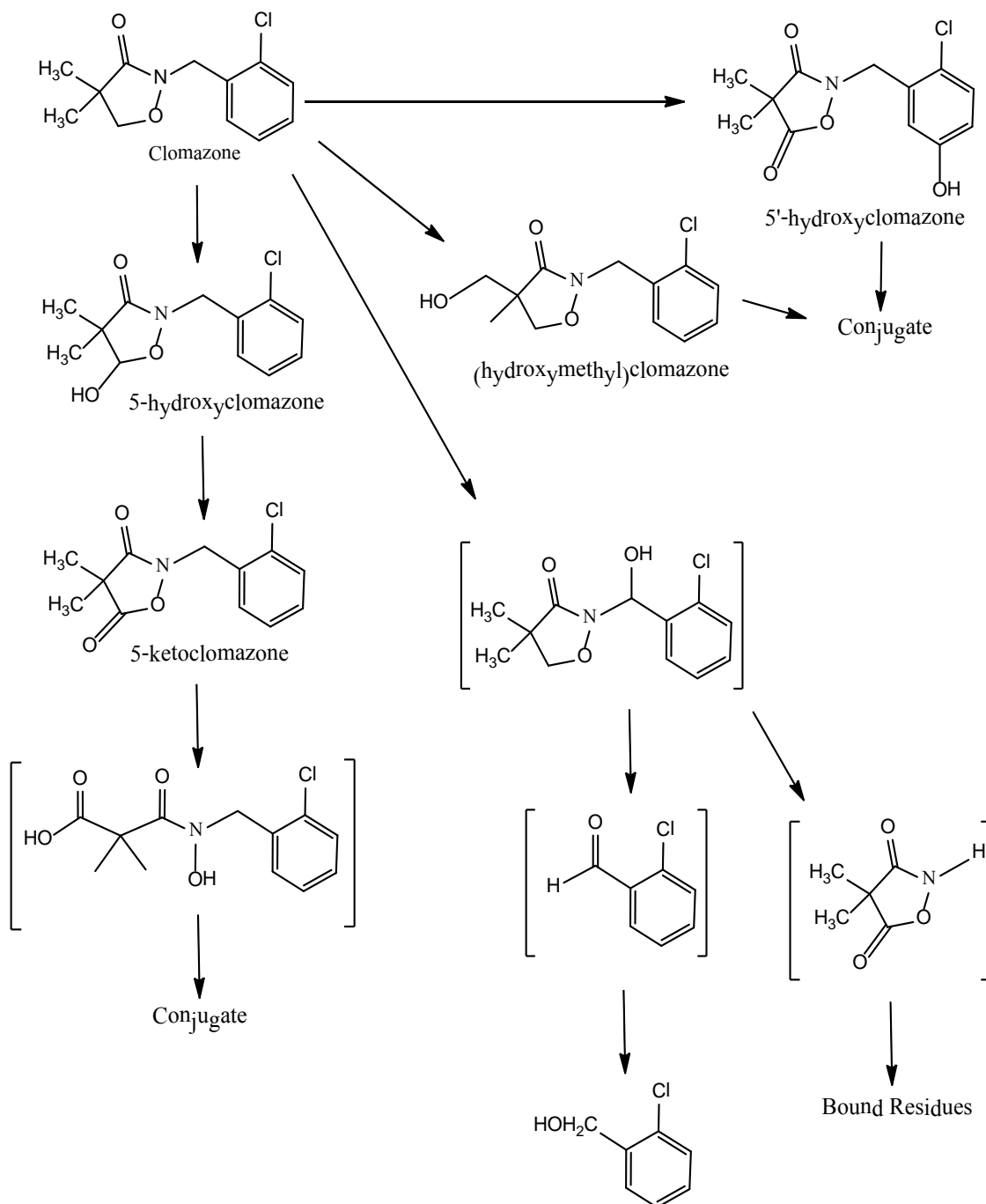


Fig. 5 Proposed metabolic pathway for clomazone in soybean plants (adapted from ElNaggar et al. 1992)

Table 1 Physicochemical properties of clomazone

Physicochemical Property	Value
Chemical name	2-(2-chlorophenyl) methyl-4,4-dimethyl-3-isoxazolidinone
Chemical Abstract Service registry number (CAS#)	81777-89-1
Molecular Formula <sup>b</sup>	C <sub>12</sub> H <sub>14</sub> ClNO <sub>2</sub>
Molecular weight <sup>b</sup>	239.7
Boiling point (°C) <sup>b</sup>	275.4 – 281.7
Water solubility at 23°C <sup>c</sup>	1100 mg/L
Vapor pressure at 25°C (mmHg) <sup>c</sup>	1.44 x 10 <sup>-4</sup>
Density at 22°C (g/mL) <sup>c</sup>	1.16
Octanol-water partition coefficient (log K <sub>ow</sub> ) <sup>c</sup>	352 (2.55)
Adsorption coefficient (K <sub>d</sub> ) <sup>a</sup>	1.5-7.4
Organic carbon normalized partition coefficient (K <sub>oc</sub> ) <sup>a</sup>	139-608
Henry's law constant (atm-m <sup>3</sup> /mol) <sup>b</sup>	4.14 x 10 <sup>-8</sup>

<sup>a</sup>Data from US EPA (2007), <sup>b</sup>Data from Tomlin (2000), <sup>c</sup>Data from CDPR (2003)