Canadian Water Quality Guidelines for the Protection of Aquatic Life

DICAMBA

Dicamba ($C_6H_6Cl_2O_3$) is a herbicide with a CAS name and number of 3,6-dichloro-2-methoxybenzoic acid and 1918-00-9, respectively. Trade names include Banvel, Dyvel, and Dycleer. Dicamba was introduced in 1967 by Velsicol Chemical Corporation for use as a herbicide and has been registered in Canada since 1963 (S. Keating, 1992, Agriculture Canada, Ottawa, pers. com.).

Dicamba is characterized by a low vapour pressure (4.5 mPa at 25ºC), high water solubility (6.5 $g{\cdot}L^{-1}$ at 25ºC), and stability to oxidation and hydrolysis under typical environmental conditions (Ashton 1982; Tomlin 1994). Dicamba has a low affinity for most soil types, having a low soil–water partition coefficient (K_d = 0–0.11 $mL{\cdot}g^{-1}$). The pKa of dicamba is 1.87, and it forms salts that are appreciably soluble in water (e.g., sodium, potassium, and dimethylammonium salts) (Tomlin 1994).

Dicamba is a selective systemic herbicide that acts as an auxin-like growth regulator used to control more than 50 varieties of annual and perennial broadleaf weeds in lawns (Tomlin 1994; Agriculture and Agri-Food Canada 1997).

Direct contamination of surface water may occur due to nontarget drift from aerial or ground boom spraying operations. Indirect contamination can occur because of runoff from treated areas or leaching into groundwater and subsequent recharging of surface waters. Extreme contamination may result from pesticide spills, deliberate dumping of tank residues, or from equipment-washing operations (CCME 1993).

Contamination of Canadian freshwater sources has been detected in the western provinces, Ontario, and Quebec, where dicamba is primarily used. Detected values range from 0.05 to 517 $\mu g{\cdot}L^{-1}$ (Frank et al. 1987; Waite et al. 1995).

Microbial degradation appears to be the most important process governing the removal of dicamba if released to water. Photolysis, hydrolysis, volatilization, adsorption to sediment, and bioconcentration are not expected to be significant removal processes (CCME 1993). The half-life of dicamba was found to be <7 d, and the substance was completely dissipated in 40 d (Scifres et al. 1973).

Based on its high solubility in water, low K_d value, and low log octanol–water partition coefficient (log K_{ow} = 0.477) (Rao and Davidson 1980; Hansch 1985), it is not likely that significant amounts of dicamba would adsorb onto aquatic sediments. It is also not likely that dicamba would accumulate to a significant extent in the tissues of aquatic organisms. Some accumulation of dicamba in freshwater algae (BCF ~10) was reported in a 32-d microcosm study where concentrations of dicamba in water averaged 0.166 $mg{\cdot}L^{-1}$, but levels in organisms higher in the food chain where negligible (Yu et al. 1975).

Dicamba is also relatively stable to degradation by hydrolysis in water. Chau and Thompson (1978) observed no detectable degradation of dicamba in either distilled water or in natural lake water over 40- and 50-d periods, respectively. Similarly, Scifres et al. (1973) reported minimal losses (5%) of dicamba over 133 d.

Water Quality Guideline Derivation

The interim Canadian water quality guideline for dicamba for the protection of freshwater life was developed based on the CCCME protocol (CCME 1991).

Freshwater Life

Data exist for eight species of freshwater fish representing five families, six of which are native to Canada. Among these species, acutely (24–96-h) toxic concentrations (LC_{50}s) of dicamba ranged from 28 to 516 $mg{\cdot}L^{-1}$ (Bohmont 1967; Johnson 1978; Johnson and Finley 1980). These data indicate that dicamba is relatively nontoxic to freshwater fish.

Table 1. Water quality guidelines for dicamba for the protection of aquatic life (CCME 1993).

Aquatic life	Guideline value ($\mu g{\cdot}L^{-1}$)
Freshwater	10*
Marine	NRG†

*Interim guideline.
†No recommended guideline.

Data on the acute toxicity of dicamba were available for nine species of freshwater invertebrates, representing five families native to North America. The amphipod *Gammarus lacustris* was the most sensitive species tested, with a 96-h LC_{50} of 3.9 mg·L^{-1} (Sanders 1969). The water flea *Daphnia pulex* was also relatively sensitive to dicamba (48-h LC_{50} = 11 mg·L^{-1}) (Sanders and Cope 1966; Hulbert 1975). It was not possible to rate the sensitivities of the other invertebrate species tested as LC_{50} values were all greater than the highest test concentration administered. These data suggest that freshwater invertebrates are comparatively more sensitive to the toxic effects of dicamba than fish. This substance, however, should be considered only slightly toxic to aquatic invertebrate species, as it was for fish.

Data are available from a single study on the effects of dicamba on 14 species of green algae (Chlorophyceae). These data suggest that freshwater algae exhibit a wide range of sensitivities to this substance. Reported EC_{50}s for green algae ranged from 100 to >10 000 µg·L^{-1} (Cullimore 1975). The interim water quality guideline for dicamba for the protection of freshwater life is 10 µg·L^{-1} (CCME 1993). It was derived by multiplying the LOEL of 100 µg·L^{-1} for the green alga *Hormidium barlowi* (Cullimore 1975) by a safety factor of 0.1 (CCME 1991).

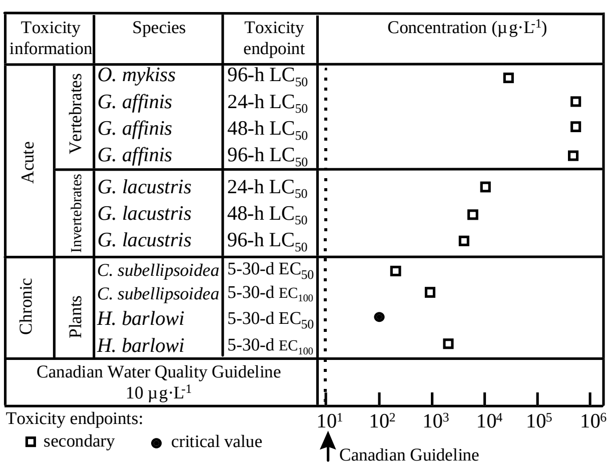

Figure 1. Select freshwater toxicity data for dicamba.

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