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► **To cite this version:**

Andy Hart, Colin Brown, Kathy Lewis, John Tzilivakis. p-EMA (II): evaluating ecological risks of pesticides for a farm-level risk assessment system. *Agronomie*, 2003, 23 (1), pp.75-84. 10.1051/agro:2002075 . hal-00885972

HAL Id: hal-00885972

<https://hal.science/hal-00885972>

Submitted on 11 May 2020

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p-EMA (II): evaluating ecological risks of pesticides for a farm-level risk assessment system

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(Received 7 February 2002; accepted 23 May 2002)

Abstract – This paper reports the development of methods for assessing the environmental risks of pesticide use in a farm-level evaluation tool. The basic risk indices that were used, and the criteria used to define different levels of risk, were designed to be consistent with national and EU regulatory risk assessment procedures. They took account of both toxicity and exposure and were responsive to the influence of local conditions and practices, such as pesticide application methods and habitat structure. They also took account of ‘higher tier’ risk assessments carried out for pesticide approvals. The outputs of the system were designed to provide clear and meaningful information for users at various levels of detail, from the individual pesticide application to an overall farm eco-rating. It is recommended that the principle of consistency with regulatory risk assessment should also be applied to the development of other policy instruments such as consumer labels and pesticide taxes.

pesticide / ecological risk assessment / farm-level assessment / software

Résumé – p-EMA : Évaluation des risques écologiques des pesticides pour un système d'évaluation du risque au niveau de l'exploitation agricole. Cet article rapporte le développement de méthodes d'évaluation des risques environnementaux liés à l'emploi des pesticides avec un outil d'évaluation à l'échelle de l'exploitation. Les indices de risque fondamentaux utilisés et les critères de définition des différents niveaux de risque, ont été mis au point pour être compatibles avec les procédures nationales et européennes. Ils tiennent compte de la toxicité et de l'exposition, et sont sensibles aux conditions et pratiques locales (méthodes d'application des pesticides, structure de l'habitat). Ils tiennent également compte des évaluations des risques « de haut niveau » conduites pour les agréments des pesticides. Les sorties du système fournissent des informations claires et significatives aux utilisateurs, à divers niveaux de détail, de l'application du pesticide à l'éco-évaluation globale au niveau de l'exploitation. Le principe de compatibilité avec l'évaluation réglementaire du risque devrait aussi être appliqué au développement d'autres instruments de politique, comme les étiquettes destinées aux consommateurs et les taxes sur les pesticides.

pesticide / évaluation du risque écologique / évaluation au niveau de l'exploitation / logiciel

1. INTRODUCTION

A computer-based environmental management tool, known as EMA [7] has been designed for farmers and their advisers to encourage more sustainable practices across the whole farm. This software uses auditing techniques to derive performance indices for a comprehensive range of farm practices, including pesticide use.

In early releases of EMA the evaluation of pesticide practices used a scoring system based on label warning phrases specified as a condition of use during the UK regulatory approvals process. The label phrases were based

largely on the toxicity of the pesticide and therefore reflected hazard rather than risk, which is a function of both toxicity and exposure. They also did not take account of many ways in which exposure could be modified by local environmental conditions and management practices.

This paper reports on the development of a new module for the EMA system, called p-EMA. p-EMA replaces the earlier hazard-based assessment with a risk-based evaluation of pesticide practices. Key priorities in developing the new module were to:

- produce indices of risk that take account of both toxicity and exposure;

Communicated by Marco Trevisan (Piacenza, Italy)

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Table I. Toxicity data used in p-EMA, showing preferred species and alternatives. Where more than one value was available for the same species, the lowest reliable value was used.

p-EMA index	Toxicity data	Species
Birds – Acute/Short term	Acute oral LD50 and 5 d dietary LC50	Lowest of bobwhite quail, Japanese quail or mallard, where not available lowest other species
Mammals – Acute	Acute oral LD50	Lowest of rat or mouse, where not available lowest species available
Fish – Acute	96 h LC50	Rainbow trout, where not available (1) Bluegill sunfish, (2) lowest other fish
Fish – Chronic	21 d NOEL	Rainbow trout, where not available (1) Bluegill sunfish, (2) lowest other fish
Daphnia – Acute	48 h EC50	<i>Daphnia magna</i> , where not available lowest species available
Daphnia – Chronic	21 d NOEL	<i>Daphnia magna</i> , where not available lowest species available
Algae – Acute	96 h ErC50	Lowest algal species available
Lemna	14 d EC50	(1) <i>Lemna minor</i> (2) <i>Lemna gibba</i>
Honeybees	Lowest of 48 h oral LD50 and 48 h contact LD50	Honeybee
Earthworms – Acute	14 d LC50	<i>Eisenia foetida</i> or lowest species available
Non-target arthropods	None (See Sect. 9)	

- make the risk indices responsive to local conditions and practices;
- be as consistent as possible with current UK and EU regulatory risk assessment procedures;
- provide indices that are capable of being aggregated across taxonomic groups, pesticides and fields to produce an overall score at farm level;
- provide indices that are compatible with the scoring system used in other parts of EMA;
- avoid requiring excessively burdensome inputs from users;
- present the results in ways that are meaningful and simple to understand.

This paper reports on ecotoxicological aspects of p-EMA: how information on exposure and toxicity were combined into risk indices and how these were aggregated to farm level. Companion papers describe the methods used in p-EMA for modelling pesticide concentrations in surface waters, ground water and soil [2], and report on case studies and the presentation of results to users [8].

2. BASIC RISK INDICES

To make the risk indices consistent with regulatory assessment, most of the calculations followed the approach set down in European regulations (Directive 91/414/EEC and its Annexes) for initial assessment of the active ingredients of pesticides. Therefore, toxicity-exposure ratios (TERs) were used for aquatic organisms, birds, mammals and earthworms, and hazard ratios were used for honeybees. The toxicity data used for p-EMA are shown in Table I.

In regulatory assessment, both acute and chronic risks are considered. However, for most groups of organisms, data for chronic toxicity are lacking for the majority of pesticides. Furthermore, there is a lack of consensus on EU regulatory

approaches to some aspects of chronic assessment (e.g. the use of time-weighted averages to allow for pesticide degradation). Therefore the current version of p-EMA considers only acute risks, except for birds where both acute and short-term risks are considered and fish and daphnia, where both acute and chronic risks are considered.

3. EFFECTS OF LOCAL CONDITIONS AND PRACTICES

A list was made of factors that could potentially affect risk (Tab. II). The basic risk indices were then adjusted to take account of the factors thought to be most important for each type of organism; details are given in the following sections. The adjustments were reasonable approximations to provide an index of risk for farmers that was responsive to local factors, but they should not be relied upon in a regulatory assessment. In some cases, potentially important factors were excluded because their influence was insufficiently understood, or because it would be impractical to provide the data necessary to estimate their effects.

4. BIRDS

4.1. Basic risk index

Two basic risk indices were used for birds: an acute TER and a short-term TER. At the end of the computations, only the lower result (higher risk) was used. TER's for sprays, treated seeds and pelleted formulations considered only dietary exposure and were calculated as follows:

$$\text{Avian acute TER} = \text{LD50} / \text{Avian acute exposure} \quad (1)$$

Table II. Aspects of local site conditions and farm practice that could potentially influence pesticide risks to non-target organisms and were considered for inclusion in p-EMA. Factors that were incorporated directly or indirectly into the calculation of p-EMA eco-ratings are marked by asterisks.

Pesticide application:	pesticide selected*
	rate of application*
	repeat applications*
	formulation type*
	time of year*
	time of day
	crop type and growth stage*
	degree of crop cover (% of ground area)*
	sprayer type/droplet size*
	implementation of no-spray zone*
	spray calibration/distribution*
	compliance with LERAPS regulations*
Weather:	quarterly average rainfall*
	annual average air temperature*
	short term weather variables – rainfall, wind speed, etc.
Site characteristics:	soil type*
	organic matter content*
	topography
	soil water status
	presence of aquifers*
	presence of surface waters*
	type of water bodies
	water flow (static vs. flowing)*
	water width and depth*
	distance to water body*
	presence and type of drains*
	presence of barriers to spray drift
	presence of honeybee hives*
	beetle banks*
	public areas
	conservation headlands*
	other wildlife habitats (e.g. wild flowers, hedges, woodland and gardens)*

* See legend.

$$\text{Avian short-term TER} = \text{LC50} / c \quad (2)$$

where

LD50 = acute oral LD50 (mg a.s. kg body·weight⁻¹)

LC50 = dietary LC50 (mg a.s. kg·food⁻¹)

$$\text{Avian acute exposure} = c \times f / w \quad (3)$$

where

f = daily food or granule consumption (wet weight, kg; see text below for derivation)

w = body weight (kg).

For treated seeds and pelleted formulations, c = nominal concentration (mg a.s.·kg⁻¹). For sprays,

$$c = a \times r \quad (4)$$

where

a = application rate (kg a.s.·ha⁻¹)

r = residue factor (mg a.s.·kg food⁻¹ per kg a.s.·ha⁻¹).

Daily food consumption was estimated using Nagy's equation for passerine birds (Note 8 of the EPPO scheme [4]) and assuming 72% moisture content for insects and 13% for seeds. Residue factors were taken from Notes 3 and 13 of the EPPO scheme [4]. Computations were based on indicator species chosen for their relevance to the use scenario. For sprays in arable crops and on orchard trees and fruit bushes, the indicator was an 11 g blue tit consuming 11 g small insects·day⁻¹; for sprays on the ground under orchards and fruit bushes, the indicator was an 18 g European robin consuming 17 g small insects·day⁻¹; for pellets and treated seeds, the indicator was a 22 g tree sparrow consuming 6.3 g seeds or pellets·day⁻¹.

Pesticide granules may be ingested by birds seeking grit. There is no generally-accepted approach for estimating the rate of granule ingestion, so one of several approaches used during recent times was adopted. Field studies in the USA [1] found a mean of 69 grit particles in the gizzards of house sparrows. For the purposes of p-EMA, it was assumed that 69 particles are ingested and lost each day, although there is experimental evidence that grit turnover in bird gizzards is more rapid so this figure may be an underestimate [5]. In a field treated with granular pesticide, such a bird might obtain its 69 particles partly as granules and partly as natural grit, depending on their relative availability and other factors such as colour and shape. Given the limited understanding of these processes at present, p-EMA made the worst-case assumption that all of the ingested particles were pesticide granules. Thus f in equation (3) was estimated as 69 × average granule weight. These assumptions were used in calculating an acute TER. Short-term TER's were not calculated for granular formulations.

4.2. Site/practice adjustments

The basic risk indices were adjusted in a number of ways to take account of local environmental conditions and management practices. The basic risk indices assumed that birds obtain all their food in the treated crop. In practice, the proportion obtained in the crop may be much lower for some individuals, especially if adjacent non-crop habitats provide attractive alternative sources of food. However, other individuals may still obtain all their food in the centre of the field. To reflect this variation in behaviour and make the risk index responsive to the presence of good wildlife habitats, two

versions of each TER were calculated: one for a bird feeding entirely in the crop ('centre-feeding bird') and the other for a bird feeding only partly in the crop ('edge-feeding bird').

For edge-feeding birds, exposure was assumed to arise partly from the crop and partly from hedges and conservation headlands exposed to spray drift. An arbitrary index of food availability was calculated, assuming that if the field is surrounded by natural habitats or gardens they would provide twice as much food for birds as the crop itself, and that the same is true for hedges and conservation headland. If the habitat in question only extended for a proportion of the field perimeter, then the amount of food it provided was reduced pro rata. Edge-feeding birds were assumed to feed in each of the three areas (crop, hedge/conservation headland, natural habitat) in proportion to its index of food availability. For example, for a field surrounded entirely by both hedges and woodland, the edge feeding bird was assumed to obtain 20% of its food in the crop, 40% in the hedges and 40% in the woodland. Pesticide residues on food in the crop were calculated as in equation (4). Food in hedges and conservation headlands was assumed to be contaminated by spray drift, so the application rate a in equation (4) was replaced by the proportion of a expected to be deposited as spray drift, averaged over the distance between the edge of the sprayed area and the centre of the field boundary (the 'drift zone'). The drift estimates were obtained using the same method used for estimating drift deposition in water bodies [2]. Thus, for a sprayed pesticide, equation (3) was replaced with:

$$\text{Avian acute exposure} = (c_{\text{crop}} \times f_{\text{crop}} + c_{\text{drift-zone}} \times f_{\text{drift-zone}}) / w \quad (5)$$

where

c_{crop} = residues on food from crop (mg a.s. · kg⁻¹)

f_{crop} = amount of food taken from crop (kg)

$c_{\text{drift-zone}}$ = residues on food from drift zone (mg a.s. · kg⁻¹)

$f_{\text{drift-zone}}$ = amount of food taken from hedges and conservation headlands (kg).

For edge-feeding birds, c in the short-term TER (Eq. (2)) was replaced by the weighted average concentration for food obtained in the three areas: crop, hedge/headland and natural habitats. For granular formulations, the proportion of grit taken from the cropped area was assumed to be equal to the proportion of food taken there. TER's for centre-feeding birds remained as in equations (1) and (2).

The overall risk indices used a weighted average of the TER's for the centre- and edge-feeding birds. The default weighting is 50:50, but the user is allowed to vary the weighting for centre-feeding birds between 30% and 100%. These weighting factors can be used to represent the proportions of local birds that predominately feed in the centre and edges of the field, or to represent the relative importance the user attaches to birds that use the centre or edge of the field. Risk is always higher for birds using the field centre, where pesticide residues are higher.

Avian exposure to treated seeds, granules and pelleted formulations is reduced if they are incorporated into the soil, and if the farmer removes any accidental spills formed during the application procedure. There are data on some of these factors [9], but they are insufficient to quantify effects on

exposure. It was therefore assumed that incorporation of treated seeds, granules or pellets would reduce exposure by a factor of 10. If there was incorporation and spill removal, then exposure was reduced by a factor of 100.

5. MAMMALS

The approach used for mammals was similar to that for birds, and only differences are described here.

5.1. Basic risk index

Only one basic risk index was used for mammals: an acute TER, calculated in the same way as the avian acute TER (see Sect. 4). Daily consumption was estimated using Nagy's equation for mammals (Note 8 of the EPPO scheme [4]) assuming 77% moisture content for short grass and 13% for seeds. For sprays in soft fruit, the indicator species was a 25 g field vole consuming 14 g short grass · day⁻¹. For sprays in arable crops, the centre-feeding indicator species was a 3.33 kg hare consuming 803 g short grass · day⁻¹ while the edge-feeding species was the field vole. For sprays in orchards, the hare was replaced with a 1.5 kg rabbit consuming 417 g short grass · day⁻¹. For pellets and treated seeds, the indicator was a 18 g wood mouse consuming 2.9 g seeds or pellets · day⁻¹.

For mammals eating short grass in arable crops, r in equation (4) was set to 112 (Note 3 of the EPPO scheme [4]). In orchards or soft fruit, allowance was made for interception when spray is applied to trees or bushes using interception estimates from the Focus groundwater report [6]. When the application is directed at the ground in these crops (e.g. applications of herbicide), there is no interception by trees or bushes.

Mammals are thought not to seek grit like birds, but to ingest pesticide granules only accidentally, e.g. adhering to food. Currently, exposure by this route is not routinely considered in regulatory assessment. However, if a single granule contains a lethal dose the p-EMA score was set to -100 (maximum risk, see Sect. 10).

5.2. Site/practice adjustments

As for birds, TERs were calculated for edge-feeding and centre-feeding mammals, and adjustments were made for the incorporation of seeds, granules and pellets, and the removal of spills. In many orchards, a strip of soil under each row of trees is kept clear of vegetation by means of repeated herbicide applications. If there are few weeds or soil is bare in this strip, then rabbits or field voles will get very little of their food there. In such cases, exposure to pesticides applied only in the strip was reduced by a factor of ten.

6. AQUATIC ORGANISMS

The regulatory risk indices for aquatic organisms are TER's, calculated as the ratio of toxicity to the predicted concentration

in surface water (maximum of spray drift or drainflow, estimated as described by Brown et al. [2]). Acute and chronic TER's were calculated for fish and daphnia, but only acute TER's were produced for algae and lemna. Acute TER's used the initial concentration in surface water, while chronic TER's used a time-weighted average concentration [2].

Local conditions and management practices that influence the amount of pesticide reaching surface waters include application rate, soil type, soil organic matter content, the degree of crop cover and a range of practices to reduce spray drift. Approaches used to model these factors are described in the companion paper [2]. If no surface water was present, no TER was calculated and all the p-EMA scores for aquatic organisms were set to zero (low risk, see later).

7. EARTHWORMS

The regulatory risk index for earthworms is an acute TER, calculated as the ratio of the LC50 to the initial soil concentration within the treated area. The approach used to model initial soil concentration is described in the companion paper [2] and took account of the application rate and the degree of interception by crop cover. In the case of orchards, interception by both the tree crop and any ground cover were considered.

Application rate and spray interception by the crop were the only local factors considered for earthworms. There are significant relationships between soil type and condition and the activity of earthworms but these are not sufficiently quantified to be modelled. No-spray zones were also ignored, because the majority of the earthworm population in the field will still be exposed. Incorporation of treated seeds, pellets or granules presumably alters the exposure of earthworms, but is also not sufficiently understood to be modelled.

8. HONEYBEES

8.1. Basic risk index

The regulatory risk index for honeybees is a hazard ratio, not a TER. The hazard ratio was obtained by dividing the application rate of the pesticide in $\text{g a.s.}\cdot\text{ha}^{-1}$ by the LD50. It was assumed that exposure of honeybees to seed treatments and granular and pelleted formulations was negligible, so for these the hazard ratio was set to zero. If the hazard ratio for a pesticide was less than 2500 but the active substance was known to have significant insect growth regulator activity, then the hazard ratio was reset to 2500 (high hazard).

8.2. Site/practice adjustments

The hazard ratio was reset to zero for applications between October and February (inclusive), when exposure of honeybees is negligible. Exposure within the treated area will be low if honeybees are not actively foraging there: this was assumed to apply if the crop itself is not flowering and there are no flowering weeds in the treated area. Cereals are

unattractive to honeybees: bees are sometimes attracted to honeydew produced by aphids, but the farmer would be unlikely to know if it were present. Beans are attractive to bees even when not flowering, because they have extra-floral nectaries.

When exposure within the crop was low but flowering plants were present in the field margin, the hazard ratio was based on exposure due to spray drift. In this case, the hazard ratio was based not on the application rate but on the proportion of the application rate expected to be deposited as spray drift, averaged over the distance between the edge of the sprayed area and the centre of the field boundary (the 'drift zone'). The spray drift estimates were obtained using the same method as was used for estimating drift deposition in water bodies [2]. When exposure was considered low both in the crop and the margin, based on the criteria described above, the hazard ratio was set to 50, or to the calculated value based on deposition in the drift zone if this was less than 50.

9. NON-TARGET ARTHROPODS

9.1. Basic risk index

The basic index used for non-target arthropods was different from all the others in that it makes no direct use of data from any regulatory studies, because most regulatory studies on non-target arthropods do not quantify the dose-response relationship. This prevented any estimation of risk reductions due to reduced application rate, and prevented any estimate of risk in the drift zone. Instead, p-EMA used a score between 0 and -100 as the basic risk index for arthropods, based on qualitative information about the activity of the pesticide. The score was set to zero for pesticides with no insecticidal activity (negligible risk), and to -50 for insecticides known to be selective (i.e. active against only a narrow range of insect taxa). Pesticides that are stated on the approved label to be suitable for use in integrated control programmes, were also assumed to be selective and given a score of -50. This score was placed in p-EMA's 'review recommended' category (see later) to encourage farmers to consider whether spraying is needed and which product is the best choice. Pesticides that are identified as being active against a broad spectrum of insect taxa will kill most of the insects in the crop and some in the margins, and were given a score of -90. Other insecticides, which are not identified as either selective or broad spectrum, were given a score of -80. A score of -80 was also given to pesticides that are not approved as insecticides but nevertheless have significant insecticidal activity, as indicated by label warnings.

9.2. Site/practice adjustments

Unsprayed buffer zones at the edge of the field are the main tool used by regulatory authorities to manage excessive risks to non-target arthropods. Therefore, if a buffer zone is required but not implemented by the farmer, the risk score was set to -100 (high risk). As a 6 m buffer has generally been considered sufficient in the UK to protect non-target arthropods from dimethoate used on summer cereals, the risk

Table III. Thresholds chosen to distinguish between the three main p-EMA score categories for each ecotoxicological risk index and groundwater concentration.

	Threshold between 'poor' and 'review recommended' (EMA score = -70)	Threshold between 'review recommended' and 'average good practice' (EMA score = -40)
General basis for threshold	Threshold used to indicate need for detailed (higher tier) risk assessment under European Directive 91/414/EEC.	Level considered indicative of reduced concern and/or reduced need for detailed assessment. Source of value is indicated in brackets.
Acute risk to birds	TER < 10	TER > 100 (91/414/EEC Annex III & EPPO[4])
Acute risk to mammals	TER < 10	TER > 100 (91/414/EEC Annex III & EPPO[4])
Acute risk to fish, acute risk to daphnia	TER < 100	TER > 1000*
Chronic risk to fish, chronic risk to daphnia, risk to algae, risk to lemna	TER < 10	TER > 100*
Acute risk to honeybees	Hazard quotient > 50	Hazard quotient < 5*
Acute risk to earthworms	TER < 10	TER > 100 (EPPO[3])
Concentrations in groundwater	Concentration > 0.1 mg·m ⁻³	Concentration < 0.01 mg·m ⁻³ *

* Where no suitable 'low risk' threshold is suggested by EU regulations or the EPPO schemes [3, 4], the interval between the 'low risk' and 'high risk' thresholds is set to a factor of ten (based on the established intervals for birds, mammals and earthworms).

score was reset to -70 (the threshold for 'acceptable' risk, see later) for any insecticide with a buffer of at least 6 m. There is no equivalent basis for evaluating buffer zones between 0 and 6 m, so these were ignored. The degree of spray drift is greater for airblast applications in orchards, so the width of buffer required to qualify for a score of -70 was 18 m rather than 6 m.

For dimethoate, an arthropod buffer zone is only required in the UK for applications after 1 April. This implies that even for broad spectrum insecticides, applications prior to 1 April can be regarded as posing acceptable (but not necessarily negligible) risk. No end date for the 'risky' period is specified for dimethoate, but it seemed reasonable to set it to harvest, since at this point field arthropod populations will suffer major disruption anyway.

'Beetle banks' are sometimes used as a means of providing harbourage for non-target arthropods and promoting population recovery after pesticide applications. Where these were present, the risk score was increased by 25. This meant that scores of up to -90 (broad spectrum) were adjusted to above the 'acceptable' threshold of -70, but scores of -100 (buffer zone required but not used) still scored below the threshold. The same adjustment was also used if there was 25% or more of 'conservation headland', as this habitat assists non-target arthropod populations in similar ways to beetle banks. The criterion of 25% was chosen because it was considered equivalent to a single beetle bank in a square field.

The hazard index was set to 0 for solid formulations or seeds, if incorporated at the time of application, to reflect their limited availability to arthropods.

10. CONVERSION OF RISK INDICES TO P-EMA SCORES

In general, EMA scores range from +100 (good performance) to -100 (poor). Earlier versions of EMA scored

environmental impacts of pesticide from 0 to -100, on the grounds that environmental effects of pesticides are generally adverse. Therefore, scores used by p-EMA also range from 0 to -100. In other parts of EMA, adverse scores below -20 are divided into two categories, 'action required' and 'poor' [8]. For p-EMA, it was decided to define individual scores (before aggregation) from 0 to -40 as 'average good practice', -40 to -70 as 'below average/review recommended' and -70 to -100 as 'poor'. It was considered that, when supported by explanatory information, these three risk categories would give clear and appropriate messages to farmers and other users.

The relationship between each p-EMA risk index and its corresponding p-EMA score was defined in two steps. First, two threshold values of the risk index were selected to distinguish between the three risk categories. As far as possible, these threshold values were based on thresholds currently used in regulatory assessment (Tab. III). In particular, the thresholds for the 'poor' category were based on the thresholds specified in European Directive 91/414/EEC for identifying which pesticides require detailed risk assessment. Second, a straight line was drawn through the two threshold values to define the p-EMA scores for other values of the risk indices (Fig. 1). This was done on the logarithmic scale because TER changes at low values are more biologically significant than changes of similar magnitude at higher values. The slopes and intercepts of the resulting relationships are shown for each taxon in Table IV. These lines can give risk to scores ranging from minus infinity to plus infinity, so they were truncated at 0 and -100 to be consistent with the scale defined for p-EMA. An analogous approach was applied to convert groundwater scores to p-EMA scores, and this is also shown in Tables III and IV and Figure 1. No conversion was necessary for non-target arthropods, as the risk scores developed for them were directly equivalent to p-EMA scores (see Sect. 9).

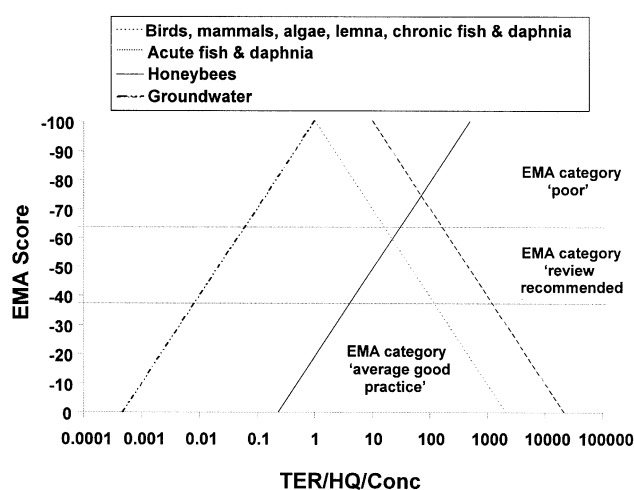


Figure 1. Graphical representation of the relationships used for converting risk indices (TERs and Hazard Quotients) and groundwater concentrations to p-EMA scores. The three main p-EMA categories are also shown. Each line passes through the respective 'high' and 'low' risk thresholds identified in Table III. The slopes and intercepts of each line are shown in Table IV. No conversion is necessary for non-target arthropods (see Sect. 9).

11. USE OF HIGHER-TIER INFORMATION

TERs are a crude measure of risk, used for initial screening in regulatory assessments. Where the initial TERs indicate a potentially significant risk, the regulatory assessment is refined, often with the use of additional 'higher tier' data that should provide a more accurate indication of risk. It was therefore important to incorporate higher-tier information into p-EMA. In particular, a pesticide should not be given a 'poor' score by p-EMA if higher-tier regulatory assessment has demonstrated acceptable risk.

It would not be practical to use higher-tier information directly in p-EMA, due to the effort required to extract the relevant data and the variable nature of higher tier studies.

Therefore, the regulatory status of each pesticide use was used to determine whether the risks have been assessed as acceptable. If so, then any risk indices that would otherwise be classified as 'poor' were reset to take them just out of the 'poor' category. By this means, the effect of any higher tier

data was taken into account. A potential disadvantage is that any risk-benefit judgements that influenced regulatory assessment become incorporated into the p-EMA scores. This was considered less important than the advantages of being able to take account of higher-tier information.

For this approach to be credible, it must be based on full and impartial assessment to modern scientific and regulatory standards. Therefore, p-EMA applied higher tier adjustments to the risk score only if at least one of the following 3 conditions was met:

1. the active substance had achieved Annex I listing as a *new* substance and the specific use was approved in the UK, or
2. the active substance had achieved Annex I listing as an *existing* substance and the specific use was listed in Section 3 of an EU review report and this use was not subject to any 'particular conditions' relating to ecotoxicological effects or groundwater in paragraph 6 of the EU report, or
3. a review of the uses of the active substance involved had been completed under the UK Plant Protection Products Regulations,

and the pesticide application in question complied with any specific approval conditions (e.g. buffer zones). Where there are 'particular conditions' in Section 6 of the EU review and criterion 3 was not met, then p-EMA applied adjustments only for organisms not mentioned in the 'particular conditions'.

Some of the factors used by p-EMA to adjust for local conditions and practices may have formed part of the basis for higher tier regulatory assessments. To avoid double-counting these factors in p-EMA, adjustments for higher-tier information were made only if the risk index remains in the 'poor' category after local site/practice factors have been accounted for. p-EMA made higher-tier adjustments only for pesticides with risk indices in the 'poor' category. In theory, risk indices for other pesticides may over-estimate risk, but there was no practical way to correct for this, because higher-tier regulatory assessments are not done for pesticides that do not breach regulatory triggers.

12. AGGREGATION OF SCORES

The risk indices within p-EMA were calculated at the level of individual applications of active substances. Actives

Table IV. Slopes and intercepts of the relationships used for converting risk indices and groundwater concentrations to p-EMA scores, shown graphically in Figure 1. Values outside the range of permissible p-EMA scores (0 to -100) are set to the limit values shown in the two columns at the right of the table. No conversion is necessary for non-target arthropods (see Sect. 9).

	'Low risk' threshold	'High risk' threshold	Slope	Intercept	Value giving p-EMA = 0	Value giving p-EMA = -100
Acute risk to birds, mammals, algae, lemna & earthworms, chronic risk to fish & daphnia	100	10	30	-100	2154	1
Acute risk to fish & daphnia	1000	100	30	-130	21544	10
Acute risk to honeybees	5	50	-30	-19.0	0.23	500
Concentrations in groundwater	0.01	0.1	-30	-100	0.00046	1

applied together in mixtures were handled independently, as if they were applied separately. However, the primary level of output to the user is at field and farm levels [8]. p-EMA scores were therefore aggregated by unweighted averaging, first across taxonomic groups, then across active substances and finally across fields. The overall score at farm level was interpreted as an index of environmental performance or ‘eco-rating’ for the average field.

Averaging scores has the disadvantage that individual applications with very unfavourable scores can be concealed from the user by being aggregated with many more favourable applications. To address this, p-EMA displays messages and icons to ensure that applications with poor individual risk scores are brought to the attention of users [8]. In addition, it was decided to apply thresholds of -20 and -60 to classify averaged scores (as in the rest of EMA), rather than -40 and -70 which are used only for classification of individual scores for particular pesticide applications and taxonomic groups (for further details, see Lewis et al. [8]). Finally, p-EMA also reports the total number of applications per field and shows how this compares to the national average for the relevant crop [8].

13. WORKED EXAMPLE

The operation of the system was illustrated using a simple example comprising a single field sown to sugar beet. The 12 ha field was on a sandy soil in the Bawtry area, near Doncaster, UK. This area has high vulnerability for leaching to groundwater, an average annual rainfall of 680 mm and average winter recharge of <125 mm. A river 2 m wide and

0.6 m deep ran along one side of the field and a further 60% of the field perimeter was hedgerow with no conservation headlands or other special habitats. The river and hedgerow were 1 and 2 m from the edge of the cropped area, respectively. The field received applications of four pesticides as three products including a seed treatment. A strip of crop 1 m in width around the crop perimeter remained untreated for all spray applications. At the time of each pesticide application, no flowering crop or weeds were present in the crop or field margin and bees were not actively foraging in the crop. Details of the applications and derivation of the environmental concentrations in soil and water are presented in the companion paper [2]. p-EMA weightings for edge- and centre-feeding birds and mammals were set at the default values of 50%. Toxicity data used in the worked example are shown in Table V and the resulting risk indices and p-EMA scores are shown in Tables VI and VII.

The overall p-EMA eco-rating score of -8 fell in the band ‘Good practice’, as do the average scores of each individual application (Tab. VII). None of the p-EMA scores for individual taxonomic groups fall below -70 , so (with the default system settings) no ‘alert’ icons were displayed. Three of the individual scores fell in the ‘Review’ band, between -40 and -70 . Communication of p-EMA outputs to the user is discussed further in a companion paper [8].

14. CONCLUSIONS

The algorithms developed for p-EMA replaced the hazard-based approach in earlier versions of EMA with a risk-based approach to the environmental evaluation of pesticide use. The

Table V. Ecotoxicity and risk data used in worked example.

	units	carbendazim	flusilazole	lenacil	thiram
Mammals – Acute LD50	mg a.s.·kg bwt ⁻¹	15000	674	11000	560
Birds – Acute LD50	mg a.s.·kg bwt ⁻¹	5826	1584	2300	2800
Birds – Dietary LC50	mg a.s.·kg diet ⁻¹	*	1584	2300	3950
Earthworms – Acute LC50	mg a.s.·kg soil ⁻¹	6	748	10000	540
Algae – Acute ErC50	mg a.s.·l ⁻¹	1.3	6.6	0.012	1.0
Lemna EC50	mg a.s.·l ⁻¹	*	*	*	*
Daphnia – Acute EC50	mg a.s.·l ⁻¹	0.17	3.4	8.4	0.006
Fish – Acute LC50	mg a.s.·l ⁻¹	0.83	1.2	2	0.048
Honeybees – LD50	ug a.s.·bee ⁻¹	50	150	25	7.9
Insect growth regulator		No	No	No	No
Fish – Chronic NOEC	mg a.s.·l ⁻¹	*	*	2.3	*
Daphnia – Chronic NOEC	mg a.s.·l ⁻¹	*	*	*	*
Insecticidal activity		No	No	No	No
Higher tier assessment**		No	No	No	No
Buffer zones		No	No	No	No

* No data available in p-EMA.

** see Section 11 for criteria.

Table VI. Risk indices (toxicity-exposure ratios and hazard quotients) for worked example, calculated by p-EMA and adjusted for local conditions as described in the text. There are no entries for non-target arthropods as their eco-scores are derived by other means (see text and Tab. VII). ‘Unknown risk’ indicates that toxicity data is lacking (see Tab. V). Arbitrary values of 100 000 are returned for aquatic risk indices in the case of seed treatments, where exposure via ‘drift’ is assumed to be negligible. See text for further explanation.

	carbendazim	flusilazole	lenacil	thiram
Mammals – Acute	6492	146	2221	67
Birds – Acute/short term	3836	521	707	76
Earthworms – Acute	274	17078	42614	48795
Algae – Acute	9705	24636	42	100000
Lemna	unknown risk	unknown risk	unknown risk	100000
Daphnia – Acute	1269	12691	29261	100000
Fish – Acute	6196	4479	6967	100000
Honeybees	0	0	0	0
Fish – Chronic	unknown risk	unknown risk	10112	100000
Daphnia – Chronic	unknown risk	unknown risk	unknown risk	100000

Table VII. p-EMA eco-scores for worked example. Groundwater scores are based on predicted groundwater concentrations (see accompanying paper [2]). The overall eco-score averaged across taxa and pesticides is –8. The p-EMA performance banding based on this overall score is ‘Good practice’.

	carbendazim	flusilazole	lenacil	thiram
Mammals – Acute	0	–35	0	–45
Birds – Acute/short term	0	–18	–15	–44
Earthworms – Acute	–27	0	0	0
Algae – Acute	0	0	–51	0
Lemna	unknown risk	unknown risk	unknown risk	0
Daphnia – Acute	–37	–7	0	0
Fish – Acute	–16	–20	–15	0
Honeybees	0	0	0	0
Groundwater	0	0	–18	0
Fish – Chronic	unknown risk	unknown risk	0	0
Daphnia – Chronic	unknown risk	unknown risk	unknown risk	0
Non-target arthropods	0	0	0	0
Average for all taxa	–9	–9	–10	–8
Alerts	None	None	None	None

basic risk indices, and the criteria used to define different levels of risk, were consistent with UK and EU regulatory risk assessment procedures. They took account of both toxicity and exposure, and were responsive to local conditions and practices. The outputs were designed to provide clear and meaningful information for users at various levels of detail, and to be compatible with the scoring systems used in other parts of EMA.

Farmer evaluation tools are one of many tools that may be used to manage pesticide risks. Other possible tools include risk indicators for use on larger geographic scales (e.g. regional or national), tools for target-setting and policy evaluation, fiscal instruments (e.g. pesticide taxes) and consumer labelling. In principle, progress towards risk reduction will be

facilitated if these tools operate in concert with one another and with the regulatory approvals process. Furthermore, as the regulatory assessment is necessarily a detailed one, it will be more efficient if other risk management tools can use information from the regulatory assessment rather than duplicate it. This principle has been applied in p-EMA and also in a national indicator for pesticide risks to aquatic biodiversity recently recommended for use in the UK [10]. It is therefore recommended that the same principle should be applied to the development of other risk management tools. Consideration could also be given to making direct links between the regulatory approvals process and the administration of other risk management tools, so that the latter can be updated promptly and efficiently when new pesticide uses are approved and old ones are modified or withdrawn.

Acknowledgement: This project was funded by the UK Department of Environment, Food and Rural Affairs (DEFRA). We thank members of the project steering committee for guidance: Dave Arnold (formerly Crop Protection Association), Mark Clook (Pesticides Safety Directorate), Dr. David Cooper (DEFRA). Views expressed in this paper are those of the authors and not necessarily those of the funding or supporting organisations. This paper is published with the permission of British Crown (Copyright 2002).

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