Existing Chemical
Secondary Notification
Assessment Report NA/899S

Australian Government
Department of Health
National Industrial Chemicals
Notification and Assessment Scheme

Phoslock™

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NATIONAL INDUSTRIAL CHEMICALS NOTIFICATION AND ASSESSMENT SCHEME
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Preface

This assessment was carried out under the National Industrial Chemicals Notification and Assessment Scheme (NICNAS). This Scheme was established by the *Industrial Chemicals (Notification and Assessment) Act 1989* (the Act), which came into operation on 17 July 1990.

The principal aim of NICNAS is to aid in the protection of the Australian people and the environment by assessing the risks of industrial chemicals and providing information to promote their safe use.

NICNAS assessments are conducted in conjunction with the Australian Government Department of the Environment, which carries out the environmental assessment for NICNAS.

NICNAS has two major assessment programmes: the assessment of human health and safety and environmental effects of new industrial chemicals prior to importation or manufacture; and the assessment of chemicals already in use in Australia to address specific concerns about their health and/or environmental effects.

Chemicals that have been assessed as new or existing chemicals may require a reassessment of the risk of the chemical under the secondary notification provisions of the Act.

This assessment report has been prepared by the Director of NICNAS, in accordance with the secondary notification provisions of the Act. Under the Act, manufacturers and importers of the chemical are required to notify the Director of new information and apply for assessment. New information can include an increase in quantity imported, the commencement of Australian manufacture, increased environmental exposure, and/or additional information becoming available on the hazardsof the chemical.

On completing a secondary notification assessment, the Director of NICNAS, in accordance with the Act, causes a draft report of the assessment to be prepared and makes it available to the applicants for factual corrections and to the public (including applicants and other interested parties) for comments. This consultation process for PEC thus includes two stages: each allows a statutory 28-day timeframe for the applicants to notify the Director of any errors and the public to submit any requests for variations of the draft report. Where variations are requested, the Director's decision concerning each request is made available to each respondent and to other interested parties (for a further period of 28 days). Notices in relation to public comment, and decisions made, are published in the *Commonwealth Chemical Gazette*.

In accordance with the Act, manufacturers and importers wishing to introduce the chemical in the future need not apply for assessment. However, manufacturers and importers need to be aware of their duty under Section 64 of the Act, including a requirement to provide any new information to NICNAS.

Assessment reports for secondary notification are available on the NICNAS website. Hard copies are available (free) by contacting NICNAS at:

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Overview

Background to the Secondary Notification Assessment

Bentonite, lanthanian (Trade Name: Phoslock™), CAS No. 302346-65-2, was assessed by NICNAS in 2001 as a new chemical, Lanthanum Modified Clay (NA/899), and is now listed in the Australian Inventory of Chemical Substances (AICS). Phoslock™ is a reaction product of bentonite clay and lanthanum chloride in which the proportion of exchangeable cations (mainly sodium) is replaced by lanthanum cations through electrostatic binding. Phoslock™ is designed to adsorb oxyanions, predominantly phosphate, from a variety of natural aquatic environments notably in order to reduce the incidence of algal blooms.

In 2007, additional data on Phoslock™ became available that warranted secondary notification. Phoslock™ was originally formulated as aqueous slurry manufactured in Australia and is currently being imported from China in a dry, granular form. The applicant now proposes to treat both recreational and drinking water reservoirs rather than restricting use to recreational waters. In addition, a number of dead fish were found floating in the Deep Creek Reservoir in southern New South Wales (NSW) following an application of Phoslock™, that was atypical in relation to both dose and addition of other agents[1]. Furthermore, a large amount of new scientific literature on environmental hazards of Phoslock™ became available. This secondary notification assessment, under the secondary notification provisions of the *Industrial Chemicals (Notification and Assessment) Act 1989* (the Act), focuses on the new data.

Manufacture/importation and uses

The original assessment indicated that Phoslock™ was manufactured in Australia formulated as a fine particulate aqueous suspension or slurry at 10-30%. Phoslock™ is presently not manufactured in Australia but is imported as a granulated solid containing >90% of the lanthanum-modified bentonite.

Import volume of Phoslock™ decreased from 2006-07 (294 tonnes) to 2009-10 (none imported) with recent imports of 102.9 tonnes in 2010-11 and no imports anticipated in 2011-12.

Phoslock™ is not available to the Australian public and is now used primarily for the treatment of large water bodies. The state or territory environmental agencies and/or appropriate regulatory authorities responsible for water quality are to be informed prior to each application.

Health effects

No toxicology data were available for Phoslock™ either during the assessment as a new chemical or for this secondary notification assessment. Published studies performed on the constituent chemicals, lanthanum chloride and bentonite, were analysed as surrogate data in the new chemical assessment of Phoslock™. Additional information on other lanthanum compounds such as the insoluble lanthanum carbonate and soluble lanthanum salts was also considered.

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[1] See Section 6.2.5 below which further discusses the Deep Creek Reservoir application.

In the current assessment, toxicokinetic data for lanthanum is based on the pharmacological use of insoluble lanthanum carbonate and studies using soluble lanthanum salts. Regardless of the source of lanthanum, the systemic toxicological effects are mediated by lanthanum ions (i.e. soluble lanthanum). In humans and animals, lanthanum from lanthanum carbonate is very poorly absorbed from the gastrointestinal tract and is eliminated predominantly via the biliary route. After repeated oral administration of lanthanum, the majority remains in the gastrointestinal tract (animals) and absorbed lanthanum accumulates in the liver (animals) and bone (animals and humans). For soluble lanthanum salts in animals, uptake of lanthanum after ingestion is higher than for the carbonate while negligible dermal absorption occurs.

Bentonite is not known to cause either acute or chronic toxicity when ingested but, similar to other fine dusts, may cause respiratory conditions when inhaled repeatedly. No acute oral, dermal or inhalation toxicity data were available for lanthanum carbonate, lanthanum chloride, and lanthanum nitrate.

Repeated oral exposure to lanthanum chloride in the rat, mouse, and dog is consistently associated with adverse effects on the stomach and liver. Liver toxicity is also observed after repeated intravenous administration to dogs.

Several in vitro (lanthanum carbonate) and in vivo (lanthanum carbonate and lanthanum chloride) genotoxicity studies are available. The weight of evidence indicates that lanthanum is unlikely to be genotoxic in vivo.

Critical studies of up to six months lanthanum chloride exposure of rodents support a no-observed-adverse-effect-level (NOAEL) for neurodevelopmental effects from oral administration of lanthanum chloride in the dose range of 0.1 – 11 mg/kg bw/day. Neurobehavioural deficits, a reduction in brain cell numbers and brain biochemistry effects are observed with a NOAEL of 2 and 0.1 mg/kg bw/day for adults and children, respectively. These NOAELs are taken forward in the estimation of Phoslock™ risks.

Occupational exposure and health risk

Phoslock™ is imported in sealed bags within shipping containers, and workers involved in transport and warehouse storage may be exposed in the event of an accident or spill.

Field workers would have the highest level of exposure to the chemical during debagging, manual transfer to hopper/conveyer, slurrying and surface application. Inhalation and dermal contact are the most likely routes of exposure with ocular exposure also possible due to accidental splashes.

Large-scale applications are intermittent and conducted over a period of days so the risk to the field workers is expected to be low. It was reported that all field workers wear facemasks, gloves, coveralls and eyewear when handling the chemical which will further reduce exposure to dust and aerosols during field operations.

Since the potential for occupational exposure is low, the risk to workers associated with the use of Phoslock™ is considered negligible.

Public exposure and health risk

Public exposure to Phoslock™, and soluble lanthanum leached from the chemical, may occur from secondary sources on the use of the chemical in recreational waterways and drinking water reservoirs. As there is an intention to use Phoslock™ to treat drinking water columns, potentially the greatest source of exposure to consumers (adults and children), and thus the risk, is likely to be oral through ingestion of tap water containing soluble lanthanum. However, dermal, ocular, and accidental ingestion exposure may also occur by contact during bathing and swimming, particularly in the treated source waters.

Data from repeat dose and developmental studies in animals suggest that there is a risk to public health from exposure to lanthanum released from Phoslock™ application in drinking water reservoirs. The risk of chronic health effects from lanthanum in drinking water is a potential concern since lanthanum has been shown to accumulate in the liver (animals) and bone (in animals and humans) and that the extent and potential adverse consequences of lanthanum accumulation in humans is unknown.

The NOAEL of 0.1 mg/kg bw/day is used in this report is consistent with the NOAEL utilised by the National Health and Medical Research Council (NHMRC) in deriving for a guideline value for lanthanum in the then proposed Australian Drinking Water Guidelines (ADWG) (NHMRC, 2010). NICNAS and the NHMRC reached the same conclusions regarding the potential neurotoxic and neurobehavioural effects of soluble lanthanum. Converting the NOAEL to a drinking water guideline value using a standardised approach gives a value of 0.002 mg/L. This concentration has not been endorsed by the NHMRC but is used as an example risk assessment to evaluate the possible health impacts for a scenario in which the concentration of lanthanum in tap water is considered to be well controlled. Another scenario in which it was assumed the lanthanum concentration in drinking water was not well controlled (0.033 mg/L) has also been evaluated.

The modelled drinking water scenarios include drinking water intake where:

- a control on the level of lanthanum existed, for which the margin of exposure (MOE) was acceptable for both adults and children; and
- the lanthanum levels are well above a controlled concentration of 0.002 mg/L for which the MOE was unacceptable to children.

These risk outcomes demonstrate that it is important to effectively manage the lanthanum levels in drinking water of the drinking water supply bodies treated with Phoslock™.

Environmental effects

Ecotoxicity data are available for Phoslock™ and soluble lanthanum salt solutions during the assessment as a new chemical and for this secondary notification assessment.

The bioavailability of lanthanum from Phoslock™ is mainly due to the presence of the free or ionic lanthanum (La^{3+}). If the free La^{3+} is not bound to any ions in solution, it is available for uptake by aquatic organisms.

The toxicity tests considered the effects of specific dose rates of Phoslock™ or lanthanum leachates/solutions observed from laboratory experiments wherein the levels of dissolved lanthanum for the treatments are taken into account when available. Aquatic species from three trophic levels were tested for mortality, immobilisation, growth, and/or reproduction. Sediment-dwelling organisms were also tested. The toxic effects associated with Phoslock™ are likely due to the bioavailable lanthanum released to overlying water as the surface applied solutions settle through the water column to the sediment. However, the dissolved lanthanum analysed in the tests does not necessarily correspond to the ionic or bioavailable La^{3+} found in solution.

Significant variations in toxicity are observed even in tests on the same aquatic species which could be dependent on parameters other than the levels of dissolved lanthanum available for uptake. Water hardness from the various water media is a significant factor in the toxicity differences. Dissolved or ionic lanthanum is less acutely toxic to aquatic invertebrates in hard waters (120 to <180 mg/L calcium carbonate ($CaCO_3$)) than soft waters (0 to <60 mg/L $CaCO_3$), due to the binding of the ionic lanthanum with carbonate ions in hard waters, as well as increased competition from multivalent hardness cations such as Ca^{2+} and Mg^{2+} for biotic ligands. In addition, experimental artefacts, such as a loss of alkalinity due to precipitation of carbonate as insoluble lanthanum salts, and unavailability

of food to aquatic organisms due to precipitation possibly contributed to the variability of the results. Some of the studies also recognised that toxic effects could be from other parameters other than direct toxicity of lanthanum.

In fish sub-acute tests, the rainbow trout *Oncorhynchus mykiss* has a median effective concentration (EC50) of 200 mg/L Phoslock™ and a no-observed-effect-concentration (NOEC) of 40 mg/L Phoslock™ in softwater. The measured dissolved lanthanum level at the EC50 and NOEC at the end of the test were 14 and 10 µg/L, respectively, but dissolved lanthanum levels at these test concentrations of Phoslock™ at any other point throughout the test are unknown.

There is a wide range of test results for aquatic invertebrates. However, the effects of lanthanum chloride show a decreasing trend with increasing water hardness (filtered tap water to hard water) in acute toxicity tests to cladocerans. The lowest effect level for lanthanum toxicity to aquatic invertebrates was in *Ceriodaphnia dubia* with measured dissolved lanthanum of 20 µg/L at EC50 of >1 mg/L Phoslock™ in softwater.

Chronic toxicity studies performed on sediment-dwelling organisms were done with sediments using natural lake/pond/spring water, with all tests conducted under hardwaters. The most sensitive species is the amphipod *Phreatogammarus helmsii* with observed acute EC50 and NOEC of 33 and <20 mg/L Phoslock™, respectively, and 6-8 µg/L measured dissolved lanthanum at the end of the test.

Although the studies aimed to determine the ecotoxicity effects of Phoslock™ or some soluble lanthanum salts, some of the authors suggested that the effects observed may be due to factors other than direct toxicity of dissolved lanthanum.

The amphipod *P. helmsii* study is used in deriving a revised predicted no effect concentration (PNEC) for ionic lanthanum based on the hierarchical framework of the *Australian and New Zealand Guidelines for Fresh and Marine Water Quality* and currently available lanthanum toxicity studies. A freshwater low reliability trigger value for ionic lanthanum is revised to 0.06 µg/L, which is comparable to a PNEC in this assessment.

Environmental exposure and risks

Lanthanum, present in Phoslock™ at approximately 5% (w/w) on a dry weight basis, is the main active ingredient which is most likely released from Phoslock™ in its dissolved or ionic form. In solution, the ionic form of lanthanum is likely to be reactive due to its high charge with the final form ultimately determined by the solution chemistry.

Since Phoslock™ is used primarily to treat environmental water bodies susceptible to blooms of cyanobacteria, the sources of environmental exposure mainly comes from direct releases into these environmental waters. The fraction of total lanthanum in Phoslock™ that is readily released in environmental waters is a key factor in determining the potential aquatic hazard of the chemical.

The highest concentrations of dissolved lanthanum occur immediately after application of Phoslock™ granules, and have ranged from 10 to 220 µg/L in overseas and Australian environmental waters. It is unknown how much of the measured dissolved lanthanum is and remains bioavailable as ionic lanthanum. The mechanisms underlying the higher than expected concentrations of dissolved lanthanum after Phoslock™ application in environmental waters are a function of the various water chemistry parameters as well as the equilibrium of ionic lanthanum binding with the available phosphorus.

Dissolved lanthanum levels do not exceed 10 µg/L one week after application as observed in the field applications of Phoslock™ granules in all the environmental water treatments reported. The exception is when there are significant rain events which could influence the

release of dissolved lanthanum possibly as a result of resolubilisation of lanthanum from complexed material in the water column or sediments.

Most of the overseas field applications of Phoslock™ adhere to the recommended dosing of 100:1 Phoslock™ to filterable reactive phosphorus (FRP). In Australia, the application in Deep Creek Reservoir (NSW) used approximately three times the recommended dose and a high level of peak dissolved lanthanum (220 µg/L) was observed. With a value of 27.6 mg/L $CaCO_3$, the reservoir was of low hardness. Dissolved lanthanum levels stabilised rapidly after a few days of Phoslock™ application.

The peak dissolved lanthanum level is the critical parameter for which acute effects can be evaluated. The most appropriate predicted environmental concentration (PEC) determination is a peak level from actual field application results of the product. For the purposes of risk characterisation, the maximum peak dissolved lanthanum concentration of 220 µg/L is reasonably representative of the worst-case peak levels achievable under any conditions of Phoslock™ field application and is taken as the PEC for acute exposure in this assessment.

The environmental risks associated with the use of Phoslock™ to prevent nuisance algal blooms in environmental water bodies include a high risk of adverse effects on aquatic organisms in some application scenarios. In these scenarios, the risk quotient (RQ) for Phoslock™ in the water column is significantly greater than one, based on the ionic lanthanum level that peaks 1-3 days after application and a conservative low reliability PNEC derived from laboratory toxicity data.

The environmental risks of Phoslock™ are essentially the environmental risks of the dissolved or ionic lanthanum it contains, and the environmental speciation, fate and toxicity of lanthanum are strongly dependent on water chemistry. The available data suggest that the environmental risks are highly site-specific. The RQ approach is based on highly conservative PEC and PNEC values derived under worst case water chemistry conditions. There are difficulties in relating risks based on laboratory toxicity data with the risk to Phoslock™ in the aquatic field environment, particularly due to differences in the form of lanthanum related to variations in water chemistry and the considerable uncertainty on factors affecting toxicity to aquatic organisms. The refinement of risk characterisations is essential where suitable models and data are available.

Recommendations

This section provides the recommendations arising from the secondary notification assessment of Phoslock™. Implicit in these recommendations is that best practice is implemented to minimise occupational and public exposure and environmental impact. The recommendations provided by the new chemical assessment (NA/899) are still applicable.

Lanthanum released from Phoslock™ is essentially available for uptake by organisms when in free ionic form not bound to any ions in solution. The human health and environmental risks arising from Phoslock™ application are dictated by the concentrations of ionic lanthanum. The level of Phoslock™ risk to humans is considered acceptable if the lanthanum levels are maintained in accordance with a controlled concentration for lanthanum of no greater than 0.002 mg/L when present in drinking water. The environmental risks of Phoslock™ are site-specific and could be minimised with adequate measures in place.

A framework for the management of risks to aquatic organisms from Phoslock™ use in water treatment is presented in the table below. Differences in physical and chemical water quality parameters, climate, geographical location, and other environmental parameters are

particularly challenging in setting a single recommendation that will capture these variations. Thus, a flexible framework approach is proposed which takes into account site-specific factors. Site-specific direct toxicity testing is recommended for Phoslock™ considering the current uncertainties on the environmental fate and toxicity of the chemical and the toxic ionic lanthanum component possibly released in aquatic ecosystems. The ideal set of circumstances that would produce minimal releases of dissolved lanthanum in environmental waters treated with Phoslock™ is used in the development of the framework for site-specific application of the chemical and may be further refined as additional relevant data are collected.

Framework for the management of risks to Phoslock™ application

The framework identifies lower and higher risk situations based on key water quality parameters and results of toxicity tests on aquatic organisms. Note that there is significant uncertainty relating to the dissolved lanthanum concentrations at which lethal or sublethal effects are observed. The following points form the basis of the proposed framework:

1. In softwater or low alkalinity water (<60 mg/L CaCO3), toxicity to sensitive species (cladocerans and fish) has been observed at comparatively low dissolved lanthanum concentrations from laboratory studies and monitoring results. Ionic lanthanum remains in solution since there is a lack of anions in the low alkaline water body to bind with and precipitate the La3+ ions to render them unavailable for uptake by organisms. The FRP level is a significant limiting factor since the applied lanthanum should be equivalent to the phosphorus available for binding.
2. In hardwater or high alkalinity water (>60 mg/L CaCO3), reduced toxicity has generally been observed, with the exception of a single amphipod study wherein minimal sublethal effects occurred at dissolved lanthanum concentrations <10 µg/L. However, there is considerable uncertainty in whether the effects observed can be attributed to dissolved lanthanum. The FRP level is less of a limiting factor since any excess ionic lanthanum not captured by the FRP would bind with the carbonate in highly alkaline waters.

The framework is presented in the table below:

Conditions	Strategy
Higher risk potential	There should be a presumption that Phoslock™ is not used to mitigate FRP. If Phoslock™ is proposed for use in these circumstances, a detailed justification should be provided prior to the conduct of the direct toxicity assessments (DTA) requirements as set out in the *Australian and New Zealand Guidelines for Fresh and Marine Water Quality*. Adherence to the recommended dosing ratio of 100:1 Phoslock™ to FRP should be followed. The following measures are recommended for conditions listed below, which are not all-inclusive, that may constitute a higher risk: **Softwater/low alkalinity (<60 mg/L $CaCO_3$) and FRP <0.1 mg/L** In all stages of the pre- and post-application monitoring, information on the key water chemistry parameters (pH, hardness/alkalinity, total phosphorus and FRP, total and dissolved lanthanum, dissolved oxygen/dissolved organic carbon (DO/DOC), chlorophyll-a) are required. The indicative DTA requirements are presented in Table 11.2. In addition, determining the porewater lanthanum concentrations, and porewater lanthanum fluxes (if any) to overlying waters, and establishing criteria to determine to what extent these posed a risk to ecosystem health are ideal. **Softwater/low alkalinity (<60 mg/L $CaCO_3$) and FRP ≥0.1 mg/L** The key water chemistry parameters (pH, hardness/alkalinity, total phosphorus and FRP, total and dissolved lanthanum, DO/DOC, chlorophyll-a) are to be measured in all stages of the pre- and post-application monitoring. The indicative DTA requirements are shown in Table 11.3.
Lower risk potential	The recommended measure for the condition that may constitute a lower risk is presented below: **Hardwater/high alkalinity (>60 mg/L $CaCO_3$) regardless of FRP** The key water chemistry parameters (pH, hardness/alkalinity, total phosphorus and FRP, total and dissolved lanthanum, DO/DOC, chlorophyll-a) are to be measured in all stages of the pre- and post-application monitoring, with indicative DTA requirements presented in Table 11.4.

Recommendations to state and territory governments and operators

Water quality parameters determine the release of dissolved (ionic) lanthanum, which pose the risks to aquatic organisms. Assessment of water quality prior to the application of Phoslock™ is essential for its safe use. The operators of Phoslock™ applications in Australian environmental water bodies (e.g. city councils, state/territory governments, applicant, and other companies possibly importing or manufacturing Phoslock™ in the future) should ensure that these evaluations are conducted prior to the use of the chemical. Pre- and post-application monitoring regimes with the analysis of key water quality parameters, as well as ecotoxicological testing of aquatic organisms, should be established and incorporated in the formal approval mechanisms required by Australian states and territories.

Phoslock™ should only be used with the prior approval of Australian state and territory agencies with responsibility for water quality management. To minimise the environmental risk from the use of Phoslock™, water quality managers in each jurisdiction should also ensure compliance with technical conditions informed by site-specific risk assessment procedures and monitoring protocols in the framework for the management of risks to aquatic organisms from dissolved lanthanum released from Phoslock™ field applications. The application of the framework should take account of the types of procedures, such as mixing methods, to be applied in the proposed field application. More importantly, the data generated from the pre- and post-application monitoring as presented in the framework could contribute to a greater understanding of the interaction of Phoslock™ to environmental waters and aquatic organisms. The dataset could be utilised to revise a more

reliable trigger value for lanthanum in line with the minimum data requirements outlined in the *Australian and New Zealand Guidelines for Fresh and Marine Water Quality*. A more reliable trigger value is essential for the improved risk characterisation and risk management of Phoslock™.

Recommendations to water utilities

After the treatment of a water body with Phoslock™, a quantity of lanthanum will be released from Phoslock™ and may exist in either the ionic or insoluble form depending on the binding with the other particulates in the water. The insoluble forms of lanthanum are not likely to persist in the water body since they are expected to be removed by sedimentation or other physical treatment processes. However, excess ionic or soluble lanthanum may be present post-application.

For environmental water bodies used as drinking water reservoirs where treatment of Phoslock™ has occurred, Australian water utility companies should monitor the levels of ionic lanthanum in drinking water and ensure that the levels do not exceed the controlled concentration of 0.002 mg/L for lanthanum.

Phoslock Water Solutions (PWS) recommend that Phoslock™ not be applied near intake towers or rising mains that supply the raw water to the treatment plant and that drinking water reservoirs treated with Phoslock™ is offline for 5-7 days to ensure the complete binding of lanthanum to phosphate and subsequent settling of lanthanum phosphate to the sediment layer. If these conditions are applied, and subsequent to normal treatment processes, achievement of an acceptable concentration in drinking water is likely in most cases without further intervention.

Secondary Notification

Under the Act, secondary notification of Phoslock™ may be required where an applicant or other introducer (importer) of Phoslock™ becomes aware of any circumstances that may warrant a reassessment of its hazards and risks. Specific circumstances include:

a. The use of Phoslock™ has changed or is likely to change significantly.

b. Manufacture of Phoslock™ has re-commenced or is likely to re-commence in Australia.

c. Phoslock™ has become available to the public.

d. Additional information has become available on the adverse health and/or environmental effects of Phoslock™.

e. Additional data have become available from the studies on the environmental fate and effects of Phoslock™ and ionic lanthanum (e.g. from studies arising from the implementation of the risk management framework for the site-specific application of Phoslock™).

The Director must be notified within 28 days of the introducer becoming aware of any of the above circumstances.

Abbreviations and Acronyms

ACT	Australian Capital Territory
ADWG	Australian Drinking Water Guidelines
AICS	Australian Inventory of Chemical Substances
ANZECC	Australian and New Zealand Environment and Conservation Council
aq	aqueous
ARMCANZ	Agriculture and Resource Management Council of Australia and New Zealand
ASTM	American Society for Testing and Materials
AUC	area under the curve
BCF	bio-concentration factor
bw	bodyweight
°C	degrees Celsius
$CaCO_3$	calcium carbonate salt
CAS	Chemical Abstracts Service
CHO	Chinese hamster ovary cells
cm	centrimetre
CO_3^{2-}	carbonate anion
CSIRO	Commonwealth Scientific and Industrial Research Organisation
d	day
DECC	New South Wales Government Department of Environment and Climate Change
dm	decimetre
DSEWPaC	Australian Government Department of Sustainability, Environment, Water, Population and Communities
DNA	deoxyribonucleic acid
DO	dissolved oxygen
DOC	dissolved organic carbon
DTA	direct toxicity assessments
EC	electrical conductivity
EC50	median effective concentration
ECL	environmental concern level
EINECS	European Inventory of Existing Commercial Chemical Substances
ERMA	Environmental Risk Management Authority New Zealand
ESRD	end-stage renal disease
FDA	Food and Drug Administration
FRP	filterable reactive phosphorus
g	gram
GLP	good laboratory practice
GHS	Globally Harmonised System of Classification and Labelling of Chemicals
H^+	hydrogen cation
$H_2PO_4^-$	dihydrogen phosphate anion
HCl	hydrochloric acid
hgprt	hypoxanthine-guanine phosphoribosyl transferase
HPLC	high performance liquid chromatography
HPO_4^{2-}	monohydrogen phosphate anion
HSIS	Hazardous Substances Information System
ICP-AES	inductively coupled plasma-atomic emission spectroscopy
ICP-MS	inductively coupled plasma-mass spectrometry

ip	intraperitoneal
ISO	International Organization for Standardization
iv	intravenous
kg	kilogram
Ksp	solubility product constant
L	litre
La^{3+}	trivalent lanthanum cation
$La_2(CO_3)_3$	lanthanum carbonate salt
$LaCO_3^-$	monovalent lanthanum carbonate anion
$(LaOH)^{2+}$	divalent hydroxylated lanthanum cation
$LaPO_4$	lanthanum phosphate salt
LC50	median lethal concentration
LD50	median lethal dose
LOAEL	lowest observed adverse effect level
LOD	limit of detection
LOEC	lowest observed effect concentration
LOQ	limit of quantification
µg	microgram
µg/L	microgram per litre
µm	micrometre
µS	microsiemens
mg	milligram
mL	millilitre
mm	millimetre
MOE	margin of exposure
mol	mole
mS	millisiemens
MSDS	material safety data sheet
mV	millivolt
NaOH	sodium hydroxide
ND	new data
ng	nanogram
NHMRC	National Health and Medical Research Council
NICNAS	National Industrial Chemicals Notification and Assessment Scheme
nm	nanometre
NOAEL	no-observed-adverse-effect level
NOEC	no-observed-effect concentration
NOEL	no-observed-effect level
NOHSC	National Occupational Health and Safety Commission
NSW	New South Wales
NTU	nephelometric turbidity units
O_2	oxygen
OECD	Organisation for Economic Cooperation and Development
OH&S	occupational health and safety
P	phosphorus
PEC	predicted environmental concentration
pKa	acid dissociation constant
PNEC	predicted-no-effect concentration
PO_4^{3-}	orthophosphate anion

ppt	parts per thousand
PPE	personal protective equipment
ppm	part per million
PQL	practical quantitation limit
PWS	Phoslock Water Solutions
QLD	Queensland
REE	rare earth element
RQ	risk quotient
+S9	with rat liver microsome preparations
-S9	without rat liver microsome preparations
SA	South Australia
SiO_2	silica
SOP	Standard Operating Procedure
SR-XRF	Synchrotron radiation X-ray fluorescence
STP	sewage treatment plant
SUSMP	Standard for Uniform Scheduling of Medicines and Poisons
TCLP	Toxicity Characteristic Leaching Procedure
TG	Test Guidelines
TWA	time-weighted average
WA	Western Australia
w/w	weight for weight
wt	weight
XRD	x-ray diffraction
XRF	x-ray fluorescence

1 Introduction

1.1 Background to Secondary Notification Assessment

The chemical Lanthanum Modified Clay (Phoslock™) (CAS No. 302346-65-2) was assessed as a new chemical under Section 32 of the *Industrial Chemicals (Notification and Assessment) Act 1989* (the Act) under the standard notification category, and is now listed on the Australian Inventory of Chemical Substances (AICS) as 'Bentonite, lanthanian'.

The New Chemicals Assessment Report (NA/899) was published in July 2001. No toxicological studies were available for the chemical. However, published mammalian toxicological studies on the constituent chemicals, lanthanum chloride and bentonite, were provided as surrogate data. Based on the data provided at the time, a hazard classification was conducted in accordance with the *Approved Criteria for Classifying Hazardous Substances* (NOHSC, 1999), and Lanthanum Modified Clay (referred to as 'Phoslock™, throughout the report) was considered a non-hazardous substance and is not currently listed in the NOHSC List of Designated Hazardous Substances contained in the Hazardous Substances Information System (HSIS) or the Standard for Uniform Scheduling of Medicines and Poisons (SUSMP).

Readers are referred to the New Chemicals Full Public Report on Phoslock™ available at the NICNAS website (report no. NA/899):
http://www.nicnas.gov.au/publications/car/new/na/nafullr/na0800fr/na899fr.pdf

Recommendations were made relating to minimising occupational and public exposure and environmental effects. These recommendations were based on the intended Australian manufacture and use of Phoslock™ to control soluble phosphate (or other oxyanion) concentrations in rivers, lakes and other water bodies.

The original environmental risk assessment of Phoslock™ carried out in 2001 concluded that it is potentially hazardous to aquatic organisms. The aquatic hazard is principally related to the potential release of lanthanum ions from this chemical, which are toxic to freshwater invertebrates in soft water. Although Phoslock™ is potentially hazardous to aquatic biota, the risks to the environment associated with the manufacture and use of the chemical were considered acceptable. This conclusion was based on studies which appeared to show that the fraction of total lanthanum in Phoslock™ which is readily released in water is low. The concentration of lanthanum ions in the water column following application of Phoslock™ to a water body was therefore not expected to be toxic to sensitive biota (NICNAS, 2001).

An assessment certificate for Phoslock™ was granted, however it was recommended that State Environmental Agencies and/or appropriate regulatory agencies responsible for water quality should be informed prior to each use of this notified new chemical. It was also recommended that sensitive benthic species such as gobies (a bottom-dwelling freshwater fish species) be protected by adequate measures, that spraying of slurries of the chemical should be avoided near banks of rivers and dams, and that a set of guidelines should be developed for the appropriate use of Phoslock™, including methods of application to water bodies. Specific aspects of the ecotoxicological profile of the chemical that warranted further investigation were also identified. These included the need for toxicity tests on burrowing amphipods and tests to determine if this clay-based material has physical impacts on fish and sediment-dwelling biota. Finally, a number of stipulations were made under Section 64 of the Act which could trigger a secondary notification assessment of Phoslock™, including reports of delayed environmental effects following application of the chemical (NICNAS, 2001).

The requirement for a secondary notification assessment of Phoslock™ was triggered following the use of the chemical to treat a bloom of cyanobacteria in the Deep Creek Reservoir that supplies potable water to the Batemans Bay/Narooma districts in southern NSW. Following application of Phoslock™ to this reservoir over a four day period between the 2nd and 5th of April 2007, a number of dead small fish were found floating in the reservoir and fish apparently continued to die for up to two weeks after application of the chemical (NSW DECC, 2007a & b; 2008a) that was atypical in relation to both dose and addition of other agents[1]. The then NSW Department of the Environment & Climate Change (DECC) also found almost no living zooplankton population in the reservoir about 7 weeks after application of the chemical (NSW DECC, 2008a). Furthermore, analyses of water samples taken from the reservoir shortly after application of Phoslock™ showed dissolved lanthanum concentrations up to 10 times greater than predicted in the original environmental risk assessment (NSW DECC, 2007a & b). The possible link between the application of the chemical and lethal effects on aquatic biota from two trophic levels, and the unexpectedly high levels of dissolved lanthanum were sufficient to require re-examination of the environmental risks associated with the introduction, use, and disposal of Phoslock™.

A number of additional laboratory ecotoxicity studies have been conducted with the formulated Phoslock™ product in the interval since the initial environmental risk assessment was performed. These additional studies have been reviewed as part of the re-assessment of the environmental risks of Phoslock™.

Phoslock™ was originally manufactured locally, formulated as aqueous slurry and intended for predominantly commercial use with a limited risk of exposure to the general public. The applicant has since advised that the chemical will now only be imported into Australia in a dry granular form. Additionally, Phoslock™ may also be used for the treatment of drinking water reservoirs. This additional information changes both the occupational and public exposure and warrant a reassessment of new human health and environmental risks.

The new data warrant reassessment, which has been carried out under Section 68A of the Act, covering secondary notifications of existing chemicals. Additionally, new toxicological data for lanthanum chloride and several studies on lanthanum, based on the US FDA-approved therapeutic use of lanthanum carbonate in controlling hyperphosphataemia in patients suffering end stage chronic renal disease (ESRD), are now publicly available. It was therefore considered appropriate to review the animal and human health studies published since the original assessment.

Data submitted for the original assessment on use, exposure, animal and human toxicity and environmental effects are summarised in this report in the relevant sections.

Details of the studies provided for assessment as a new chemical are reproduced in the Appendix. New data submitted for this assessment are discussed in detail and identified by the abbreviation **ND**.

1.2 Declaration

Declaration as a secondary notification was initiated when NICNAS received new data from Phoslock™ Pty Ltd regarding environmental effects of Phoslock™ that were not available during its assessment as a new chemical. These are:

1. Several aquatic biota studies.

[1] See Section 6.2.5 below which further discusses the Deep Creek Reservoir application.

2. Data describing an adverse impact on some aquatic species in a water supply body after treatment with Phoslock™.
3. Relevant human health and environmental studies on lanthanum published since 2001.
4. Information on an intended new use of Phoslock™, namely, to drinking water supplies.

A notice was published in the *Chemical Gazette* of 2 August 2007 requiring all persons who introduce Lanthanum Modified Clay into Australia either by manufacture or import, to apply for secondary notification.

1.3 Objectives

The objectives of this assessment were to review the new information that has become available since the publication of the 2001 New Chemical Assessment Report, and, where appropriate, revise the original assessment to:

- characterise the hazards of Phoslock™ to human health and the environment;
- characterise potential occupational, public and environmental exposure to Phoslock™;
- characterise the risks of adverse effects resulting from exposure to workers, the general public and the environment; and
- make appropriate recommendations to control exposures and/or reduce potential risks for workers, the general public and the environment.

1.4 International perspective (ND)

In the United Kingdom, Phoslock™ has been evaluated by the Health and Safety Executive to be not notifiable under the *Notification of New Substances Regulations 1993* on the basis that it is a surface-modified clay and that both the substrate and the clay are EINECS-listed. This is based on a European Commission decision in 1982 stating that surface-modified articles are considered articles and the reporting should take the base substance and coating agent separately. It is also considered out of scope of the *Biocidal Products Directive (98/8/EC)*. The product has been used to reduce cyanobacterial blooms in the recreational waters of the Clatto Reservoir and Loch Flemington in Scotland with the Scottish Environment Protection Agency (SEPA) and the Centre for Ecology and Hydrology (CEH) having undertaken water quality testing before and after product application.

Phoslock™ is neither registered nor pre-registered in the European Chemicals Agency's (ECHA) Registration, Evaluation, Authorisation and Restriction of Chemical Substances (REACH). Various forms of the constituents of Phoslock™ were either registered or pre-registered.

Approvals for the use of Phoslock™ (as Bentophos®) have been granted in Germany and Poland. In Hungary, Bentophos® can be used under a biocide exemption. The product had been applied in various recreational lakes in Germany to reduce bioavailable phosphorus. In Netherlands, the product has been used in recreational waters to treat algal blooms.

In New Zealand, the applicant provided the NZ Environmental Protection Authority (then ERMA – Environmental Risk Management Authority) a self-assessment report and approval has been granted since. Phoslock™ is not classified under the NZ EPA's Hazardous Substances and New Organisms (HSNO) regulations. Phoslock™ has been repeatedly applied in Lake Okareka to reduce the phosphorus from the lake. Thorough monitoring of the lake was conducted by the regional authority responsible for the lake.

In North America, Phoslock™ has been assigned a pre-manufacture notice (PMN P-03-0313) by the United States Environmental Protection Agency (US EPA) for the chemical lanthanian bentonite for use as a binding agent for removing anionic contaminants from natural waters and industrial effluent streams. Laboratory testing was conducted in Canada

prior to field application to the Lake Simcoe watershed. Phoslock™ has been tested and granted NSF ANSI 60 certification for application to drinking water in North America (Phoslock™ Phosphate Sequestering Agent, certificate # Revision: 01/07/2013, reference WQA Web Portal, http://12.2.248.199/goldseal/detail2.cfm?tableDefID=6&companyID=6506). Phoslock Water Solutions (PWS) recently advised that Phoslock™ has been certified by the Brazilian Ministry for the Environment for import, sale and use in Brazil.

1.5 Peer review

During all stages of preparation, this report has been subject to internal peer review by NICNAS and external peer review of the environment component by Germany's Federal Environment Agency (Umwelt Bundes Amt – UBA).

1.6 Applicant

Following the Secondary Notification declaration of Lanthanum Modified Clay, one company applied for assessment of this chemical.

In accordance with the *Industrial Chemicals (Notification and Assessment) Act 1989*, NICNAS provided the applicant with a draft copy of the report for comment during the corrections/variations phase of the assessment. The applicant details are as follows:

PHOSLOCK™ WATER SOLUTIONS LTD
PO Box 521
St Leonards NSW 1590

2. Chemical Identity, Physical and Chemical Properties

2.1 Chemical identity

Chemical Name:	Bentonite, lanthanian Reaction product of bentonite or equivalent clay and lanthanum chloride, in a substitution process in which most of the sodium in the bentonite is replaced by lanthanum, with variable substitution at other sites for elements such as calcium and magnesium
Trade Name	Phoslock™
Other Names:	Lanthanum modified bentonite Lanthanum-modified clay (LMC) Rare earth-modified clay
CAS Number:	302346-65-2
Molecular Weight:	Not defined
Method of Detection and Determination:	X-ray fluorescence (XRF) and X-ray diffraction (XRD)
Degree of purity:	>95%
Additives/Adjuvants	None

2.2 Physical properties (ND)

At the time of the original notification, Phoslock™ was manufactured in Australia as a 10-30% w/w slurry in water. However, it is now manufactured in China and is formulated into a granulated solid containing >90% w/w of this chemical (IMT, 2004). This commercial end-use product has the trade name Phoslock™ granules and is the only form in which the chemical is currently imported into Australia (IMT, 2004 and PWS, 2007a).

The physical properties of Phoslock™ manufactured in China have not been reported. It is therefore not known whether further manufacturing steps involved in the production of the granular product (e.g. dewatering and granulation of the original slurry formation) has significantly changed key physical properties of this clay-based chemical including the size of its fundamental clay particles, their cation exchange capacity (CEC), and the settling velocities of the particles in water. The additional manufacturing steps could reduce the amount of unbound lanthanum ions within the Phoslock™ clay matrix and improve the handling characteristics). The granular form is now composed of >90% lanthanum modified clay. Prior to 2007, the product contained up to 5% precipitated silica dispersing agent. The physical properties of the granules are given in Table 2.1.

Table 2.1. Physical properties of Phoslock™ granules (PWS, 2006).

Property	Value
Physical state	Granular solid
Appearance	Brown free-flowing granules
Phoslock™ content	>90%
Dispersing agent	Precipitated silica (2.5-5%)
Water content	2.5-5%
Size of granules	2-4 mm × 1-3 mm
Bulk density	910-960 kg/m^3
pH (1% solution)	6.8-7.5
Dust content	<1% weight of 50 µm particles

The distribution of particle sizes in well-dispersed aqueous slurries of this product is mono-modal with a peak in the range 10-100 µm. The nominal size of particles in these slurries is in the range 1-100 µm (PWS, 2006). The proportion of these particles which are aggregates of smaller fundamental clay particles has not been reported.

PWS has provided information on the physical properties of Phoslock™ as currently used (2013). Precipitated silica has ceased to be added as a dispersing agent in the Phoslock™ formulation since 2007 as a different type of bentonite, which does not require a dispersant, is currently being utilised. In addition, the current moisture content and bulk density are 6-8% and 1050-1100 kg/m^3, respectively.

2.3 Chemical properties (ND)

Phoslock™ is a modified bentonite clay product in which a proportion of exchangeable cations (mainly sodium) are replaced by trivalent lanthanum cations (La^{3+}) (NICNAS, 2001). The lanthanum cations are derived from the readily water soluble chloride salt of trivalent lanthanum (284.5 g of $LaCl_3 \cdot 7H_2O$ dissolved in 100 g of water at 25°C; Powell & Burkholder, 1960), which is then mixed with a slurry of bentonite clay in the manufacturing process (NICNAS, 2001). The cation exchange process is achieved by electrostatically binding the lanthanum ions into the bentonite clay, substituting for the typical cations in natural bentonite (Groves, 2010).

Phoslock Water Solutions (PWS) claimed that the fraction of total lanthanum readily released from Phoslock™ in water has been reduced in the granules compared with the original slurry formulation following from changes in the manufacturing process required for production of the granular formulation. In particular, a dewatering step is used to dry the chemical for granulation, which presumably removes some of the manufacturing brine containing excess La^{3+} and therefore reduces the quantities of non-specifically bound quantities of this element in the dried and granulated end-use product (IMT, 2004).

2.3.1 Mechanisms of action (ND)

Equilibrium studies of Phoslock™ were examined by Haghseresht et al. (2009) wherein varying amounts of the granules (0.01-1 g) were added to 200 mL of 10 mg/L phosphorus solution and the resulting solution analysed for phosphate levels. From the adsorption isotherms plotted by the Langmuir equation, the absorption capacity was maximised when the equilibrium solution level was less than 1 mg/L FRP. The maximum adsorption capacity values from the equilibrium equations validated the reaction strength of a 1:1

lanthanum:phosphorus stoichiometry. The phosphate adsorption of Phoslock™ approached equilibrium in less than 1 hour, with demonstrated increased adsorption rates at higher temperatures (Haghseresht et al., 2009).

The main active ingredient of Phoslock™ is lanthanum, present in the modified clay product at approximately 5% (w/w) on a dry weight basis (Groves, 2010). Lanthanum is most likely released from Phoslock™ in the form of soluble or dissolved or ionic lanthanum. In solution, ionic lanthanum is likely to be reactive because of its high charge, for which the ultimate form is dictated by the solution chemistry. Depending on the binding of lanthanum ions with other ions in environmental waters, the resulting compound could render the lanthanum unavailable for uptake. Bioavailable lanthanum in solution is in the form of excess lanthanum ions beyond the capacity of the binding reactions in the environmental waters. Some of the lanthanum reaction mechanisms relevant to this assessment are presented below:

- La-bentonite + $M^+ \Leftrightarrow La^{3+}$ + M-bentonite (M^+=Na^+, K^+, Ca^{2+} etc)
- $La^{3+} + PO_4^{3-} \Leftrightarrow LaPO_4$
- $La^{3+} + CO_3^{2-} \Leftrightarrow La_2(CO_3)_3$
- La^{3+} + humic acid interaction
- $La^{3+} + OH^- \rightarrow (LaOH)^{2+} \rightarrow$ further hydroxo species

The 1:1 salt lanthanum phosphate ($LaPO_4$) has an extremely low solubility in water, 1. 4 x 10^{-13} mol/L, calculated from the solubility product constant (K_{sp}) of 2 x 10^{-26} mol^2/L^2 at 25°C and I = 0 mol/dm^3 reported by Liu & Byrne (1997). Similarly, lanthanum carbonate ($La_2(CO_3)_3$) also has a very low water solubility, 1.02 x 10^{-7} mol/L, calculated from the K_{sp} of 4 x 10^{-34} mol^5/L^5 at 25°C and I = 0 mol/dm^3 reported by Martell & Smith (1974). The saturated La^{3+} levels in equilibrium with solid $LaPO_4$ and $La_2(CO_3)_3$ are 1.94 x 10^{-11} and 5.81 x 10^{-5} g/L, respectively. Further interactions of lanthanum in solution are discussed in their relevant context in Section 6.2.2.

2.3.2 Additives and impurities (ND)

The manufacture of Phoslock™ involves mixing a natural clay mineral (bentonite or similar clay) with lanthanum chloride (NICNAS 2001). As the reactants used in the manufacture of this chemical are fully inorganic, the main impurities in Phoslock™ are expected to be predominantly metals and other inorganic contaminants. These can presumably include a range of other environmental cations, including those derived from rare earth elements other than lanthanum, metalloids such as arsenic, and non-metals such as phosphorus and the halides.

2.3.3 Decomposition products

The lanthanum ions in Phoslock™ have been understood to be firmly bound within the anionic inter-layer spacing of the host clay mineral phase (NICNAS, 2001). The initial release of La^{3+} cations from the clay particles into water occurs in the form of simple lanthanum salts and/or soluble lanthanum complexes. The released ionic lanthanum will ultimately form a variety of species in water (Section 2.3.1) whose structure and concentration will be determined by the site-specific chemistry of environmental waters.

The mechanisms underlying the release of lanthanum ions from Phoslock™ granules in environmental waters are currently not known. However, if release occurs by a cation-exchange mechanism, lanthanum ions should be replaced by other charge compensating cations among the basal layers of the constituent clay particles. Therefore, an important issue to resolve is the location and concentration of exchangeable cations within the

bentonite clay particles of Phoslock™ granules before and after extraction with a range of environmental water samples. This information is highly relevant to the environmental fate and effects of this chemical and requires detailed further investigation (Section 6).

2.4 Methods of detection and analysis (ND)

The routine methods of analysis for Phoslock™ granules are elemental analysis, lanthanum leach testing, FRP removal, pH of hydrated slurry, moisture content, and particle sizing. However, they are assumed to include methods for confirming that the content of Phoslock™ and the physical properties of the granules (e.g. particle size and bulk density) are within specifications.

The analyses of random samples of Phoslock™ granules taken subsequent to the field application of the chemical to the Deep Creek Reservoir in April 2007 have been presented. Although not described in detail, the results included measurements of particle size (by an unspecified technique) and extractable lanthanum, arsenic and phosphorus levels. The measurements of metal and phosphorus levels were made on water and acid (HCl, 1 mol/dm^3) extracts of the granules by means of Inductively Coupled Plasma Atomic Emission Spectroscopy (ICP-AES). The practical quantitation limit (PQL) for the metal and phosphorus in this laboratory by these methods was reported as follows: La (1 mg/kg); As (4 mg/kg); P (10 mg/kg) (PWS, 2008b).

Analysis undertaken in field and laboratory studies has reported "dissolved lanthanum". This refers to the lanthanum remaining in water following filtration through a 0.45 or 0.2 μm filter. The "dissolved lanthanum" is expected to include a number of different species of lanthanum with a range of bioavailability, including colloidal Phoslock™ and lanthanum phosphate, as well as hydroxo complexes and free La^{3+}. The speciation of "dissolved lanthanum" will vary greatly depending on the chemistry of the water to which it is applied. Unbound lanthanum ions are expected to be readily bioavailable compared to insoluble lanthanum species (e.g. lanthanum phosphate) (refer to Section 8)

3. Manufacture and Use

3.1 Manufacture and importation

The original assessment indicated the chemical was to be manufactured in Australia. Phoslock™ was formulated as a fine particulate aqueous suspension or slurry at 10-30% solid (w/w), however the applicant indicated that in future there could be other formulations of the notified chemical.

Information provided for this secondary notification indicates that Phoslock™ is no longer manufactured in Australia. From mid-2004, Phoslock™ was manufactured in China and was imported into Australia in a dry granulated form (typical purity >90 % w/w). A relatively small quantity (40 tonnes) was imported as a powder (equivalent to a pulverised granule) during the period 2005-2007 however this form is no longer used, marketed or imported and is not further considered in this assessment. Phoslock™ granules, packed in 25 kg polylined paper bags, are transported by sea and then by road freight from wharves to the importer's warehouses in 6-m shipping containers packed on pallets wrapped with shrink-wrap plastic, with each container holding approximately 20 tonnes of product. The pallets are unloaded by forklift and stored in the warehouses *in-situ*.

3.2 Quantities imported

The original assessment had estimated that up to 5000 tonnes of Phoslock™ would be manufactured each year from 2001 until 2005. Import data for Phoslock™ (Table 3.1) show that import volumes decreased with no product imported in 2009/10. Recent information from the company indicated import volumes for 2010/11 and 2011/12.

Table 3.1. Quantity of Phoslock™ imported into Australia.

Formulation	Total quantity of Phoslock™ (tonnes)							
	2004-05	2005-06	2006-07	2007-08	2008-09	2009-10	2010-11	2011-12
Granule	60	168	352	294	23	0	102.9	42
Powder	0	30	10	0	0	0	0	0
TOTAL	60	198	362	294	23	0	102.9	0

3.3 Use

The original assessment indicated that Phoslock™ was to be used to control soluble phosphate concentrations in natural aquatic systems including lakes, rivers, estuaries, dams and ornamental ponds as well as artificial environments including waste effluents. It could also be used to remove a range of other oxyanions, such as arsenate, in natural and artificial environments.

Information provided for this secondary notification states that the applicant expects to apply the product in larger water bodies which may include treatment of an estimated two major drinking water and/or recreational reservoirs in every Australian state or territory. Phoslock™ is to be applied where the aquatic systems are degraded by toxic algal blooms. Further details are discussed in Section 5.

In addition to its use in Phoslock™, lanthanum and its compounds are used in a diverse and expanding range of applications. Lanthanum is used in glass polishing, permanent magnets, lighter flints and as a catalyst in petroleum refining, automobile emission control and self-cleaning ovens (Goering, 2004). The chemical is also incorporated as an alloy in nickel-metal hydride (Ni-MH) rechargeable batteries. Lanthanum oxide is used in glass, camera

lenses and in carbon-arc electrodes in lamps used in projection lighting. Lanthanum carbonate (proprietary name: Fosrenol®) has been registered by international medical regulators and approved for therapeutic use to reduce serum phosphate in patients with end-stage renal disease (ESRD).

Bentonite has been used as an absorbent in pet litter, a lubricant in the extrusion of animal feed and oil well drilling fluids, a base in cosmetics and in the manufacture of cement and plasters (HSDB, 2007). It has been approved as a food additive in Australia and is used as an anti-caking agent (FSANZ, 2007).

4. Occupational Exposure

4.1 Sources of occupational exposure

4.1.1 Import and storage

Phoslock™ arriving on pallets in steel shipping containers to ports in Sydney, Perth and Brisbane has been transferred by road to storage facilities in the same city. Updated information (2013) indicates that shipment into Australia in recent years has been to Perth and Brisbane only. According to information provided by the applicant, the warehouse manager is the only person involved in unloading the pallets by forklift. During transfer operations exposure to the chemical is not likely except in the event of packaging breach. Damaged or leaking bags are repaired by workers wearing personal protective equipment such as coveralls and safety glasses. Spills are swept up and re-packaged where possible, however contaminated product is disposed of in standard waste bins.

4.1.2 Transport to application site

Ten to twenty transport workers transfer Phoslock™ granules from the storage warehouse to the field application site. The applicant indicated that these transport workers are commonly contract haulage staff. Each truck is loaded by the warehouse manager via forklift. In general, the individual 25 kg bags are not manually handled until they are opened at the time of application. During transfer operations exposure to the chemical is not likely except in the event of packaging breach (leaking bags). For orders of less than one tonne, the 25 kg bags are manually removed from a pallet and transferred to road transport for delivery to the application site.

4.1.3 Field workers

Prior to the commencement of a project, the applicant requires the completion of a 'Job Safety Analysis/Safe Work Method Statement' (JSA) that addresses, for each specific task, the potential hazards, degree of risk, the control measures to be implemented, and the personnel responsible for actioning them. Site induction is required before work starts, and the JSA is required to be presented and discussed at the site induction.

In large-scale applications, Phoslock™ granules in 25 kg bags will be unloaded from trucks using all-terrain forklift trucks to move pallets of bags as close as possible to the application site. They are held at a suitable height for two field staff to manually open and empty the bag into a hopper/conveyer at the shoreline. The granules are transferred via an uncovered conveyer belt to a hopper on a motorised boat. A water-based worker is responsible for the mechanised mixing of the granules with water and application of the slurry over the water surface using a continuous mix and spray venturi system. This operation can last several hours over several days. For example, at one of the larger application sites, 50 tonnes (2000 bags) of Phoslock™ were opened over a 3-day period at 6 hours/day, with 600 kg (24 bags) being emptied and loaded at a time for each boat trip. In small scale applications, two field workers will slurry the granules on shore with a motorised, continuous mix and spray system and use a hose at the shoreline to spray Phoslock™ evenly over the water surface.

4.2 Estimates of occupational exposure (ND)

The exposure for importation, storage and transport workers of Phoslock™ in bags as a granulated solid is negligible except in the event that the packaging is breached.

All of the Phoslock™ is used outdoors on water bodies creating the possibility of exposure to dust during the processes of debagging, conveyer loading and mixing. Inhalation exposure would be the predominant route of occupational exposure to workers during these activities. Minimal exposure occurs as the exposure is considered to be of short duration and intermittent in nature.

Dust can be generated when bags are opened manually and emptied into a hopper, and again when the conveyer belt empties the granules into the automated mix and spray system. Exposure to dust and aerosols by inhalation (with potential also for eye and dermal contact) for the field staff is possible during application. However, the duration of exposure is minimal and the actual application is intermittent in nature. In addition, the applicant specified the provision of protective equipment for field workers and the supplied OH&S Policy for Phoslock™ stated that protective equipment including eyewear, chemically impermeable gloves, work clothing (Breathalon suit) to cover arms, legs and torso and facemask should be used.

Boat drivers or other workers who participate in the spray application of Phoslock™ may be exposed to inhalation of dust and aerosols and also possibly eye and dermal contact with the slurry. However, this exposure will be limited by forced draught ventilation and also the short duration and frequency of applications.

From this assessment, potential for exposure of field workers to Phoslock™ is considered to be negligible.

5. Public Exposure

5.1 Exposure assessment methodology

The purpose of this evaluation is to determine the magnitude of public exposure to Phoslock™ and released lanthanum as well as the frequency and duration of that exposure. This requires an understanding of the routes by which exposure occurs, together with an understanding of the variability of consumer exposure as a result of differing use patterns and environmental conditions.

Products containing the chemical are not available to the public for domestic use. All applications of the product in Australia are conducted by personnel authorised by the applicant in accordance with individual work method statements.

Actual measured data are always preferable in an exposure assessment. Where Australian data were not available, overseas data were used. Modelled data were used when no measured data were available.

In this assessment of specific exposure pathways, the 'reasonable worst-case' approach is used in which estimates are based on worst-case, but plausible, exposure scenarios. It is believed that this approach will address practically all individuals within the target population. In addition, a 'typical' exposure estimate is performed if information is available to determine a use pattern representing an average for the target population. However, it should be noted that there may still be uncertainties associated in deriving such exposure estimates although care has been taken to address them.

5.2 Exposure estimates via secondary sources (ND)

Public exposure to Phoslock™ via secondary sources is estimated only for each of the following scenarios:

- Presence in drinking (tap) water; and
- Presence in lakes, rivers or dams used for recreational activities.

5.2.1 Estimate of lanthanum released from Phoslock™

Lanthanum in drinking water may be soluble or its presence may be associated with the insoluble particulates.

Upon addition of the Phoslock™ to the water body, it is expected that the bentonite clay particles, as the major constituent of Phoslock™, will be removed by sedimentation and other physical treatment processes. Also, a quantity of lanthanum will be released or leached from the bentonite clay and may be present as either soluble lanthanum or insoluble forms that are associated with particulates. As is the case for the clay-bound lanthanum, the latter are not expected to persist in the water body and be readily bioavailable. On this basis, primary exposure scenarios will consider soluble lanthanum (i.e. lanthanum ions).

As discussed in Section 5, the available data for the chemical do not allow the calculation of a broadly applicable concentration for soluble lanthanum released into treated water columns. The amount of soluble lanthanum released will likely depend on a number of variables such as the application rate and prevailing water chemistry factors including pH, hardness and dissolved humic material.

For example, water samples taken from Deep Creek Reservoir, New South Wales (NSW) after treatment with Phoslock™ contained up to 220 µg/L soluble lanthanum with the concentration remaining elevated at 98 µg/L lanthanum for up to 4 weeks following application. This higher than predicted leaching of lanthanum from Phoslock™ under water

conditions prevailing at Deep Creek Reservoir at the time of application has not been resolved (see Section 6.2.5 for discussion).

In another trial at the Torrens Lake, South Australia (SA), a maximum soluble lanthanum concentration of 110 µg/L was recorded at one of three sampling sites on the day of application but was reported to have decreased to <5 µg/L at all sites within 10 days. Following a rain event at 20 days post-application, the levels of soluble lanthanum had increased at the 3 sites to a maximum concentration of 24 µg/L suggesting that remobilisation of lanthanum may occur upon changes to water conditions. Measurement of lanthanum in monitored water bodies after Phoslock™ application presented in Table 5.1 has shown an initial release of lanthanum that is not well correlated with dose rate and variable concentrations that have generally exceeded background levels in the longer term.

Table 5.1. Time course of lanthanum releases from Phoslock™ in treated water bodies.

Treatment site	Date	Measured maximum soluble lanthanum concentration post-application (µg/L)			
		Day 0	1 week	4 weeks	8 weeks
Australian sites					
Torrens Lake	6-8/3/07	110	5	24	5
Deep Creek Reservoir	2-5/4/07	220*	ND	98	33
Gnowangerup Dam No. 2	30/1/08	10	<5	ND	ND
University of Queensland	1/8/06	60	30	10	<10
Overseas sites					
Rauwbraken, Netherlands	21/4/08	ND	ND	41**	4
De Kul, Netherlands	18-21/5/09	ND	ND	2	<2
Lake Niedersachsen, Netherlands	19/3/08	100	91^	14**	5
Sentosa, Singapore	1/11/06	ND	97	ND	22^^
Mill Pond, USA	22/8/11	52	26	18	<10

*Measured at 3 days post-application; **Measured at 3 weeks post-application; ^Measured at 2 weeks post-application; ^^Measured at 7 weeks post-application; ND not determined

The soluble lanthanum levels in the treated water bodies shown in Table 5.1 decrease over time (as a result of the formation of insoluble complexes with oxyanions and humic acid), however, results from Torrens Lake and Deep Creek Reservoir indicate that levels can fluctuate and be persistent.

The National Health and Medical Research Council (NHMRC) proposed in 2010 a guideline value for lanthanum in drinking water in the then Draft *Australian Drinking Water Guidelines* (NHMRC, 2010) of 0.002 mg/L (2 µg/L). The NOAEL utilised in the derivation of the draft guideline value is consistent with the most critical health effect and threshold level determined in this assessment (Section 9.2). This guideline value for lanthanum was not included in the Final ADWG in 2011 (NHMRC, 2011). For public health considerations, this value has been assigned to represent a controlled concentration of lanthanum in drinking water as provided to the consumer to examine whether the value is sufficiently protective of public health. The maximum concentration of dissolved lanthanum that was measured in the waters of Deep Creek Reservoir at 8 weeks following Phoslock™ treatment (33 µg/L) will be considered as the reasonable worst-case concentration of soluble lanthanum. While it is possible that higher concentrations of dissolved lanthanum may occur in water columns at some period after application (due to remobilisation, for example) these are considered likely to be transient excursions.

Two routes of exposure to the soluble lanthanum released from Phoslock™ are of main concern regarding its use to treat both water that may be a source of drinking water and recreational reservoirs. In drinking water, oral exposure may occur during intentional

ingestion. During swimming in recreational water bodies, dermal exposure will be dominant and oral exposure may also occur via inadvertent swallowing.

5.2.2 Drinking water

Individuals may be exposed to lanthanum in drinking water when consuming tap water as a beverage, indirectly from food and drinks containing water, or directly from swallowing the treated water or incidentally while swimming. However, the latter forms of exposure are likely to be relevant only for the limited populations swimming in the treated water bodies. It is rare that individuals will be exposed via both routes of exposure. As discussed in Section 9.4 the treatments used to prepare drinking water from the source water are expected to reduce the concentrations of lanthanum compared with the source water.

Lanthanum available for uptake from water would be the dissolved lanthanum. Lanthanum's bioavailability from Phoslock™ is mainly due to the presence of free La^{3+} ions not bound to the clay or to any ions in solution.

Exposure by ingestion is routinely estimated from the volume of water consumed by a person and the concentration of the contaminant in the water. If data are available, the fraction of the contaminant absorbed (bioavailability) in the gastrointestinal tract can be estimated; in its absence, a conservative estimate of 100% uptake is assumed.

Received oral exposures were estimated from the typical and reasonable worst-case levels of soluble lanthanum present in drinking water. The calculations were conducted for adults and for children. All the adults are considered to be one group, and sub-groups such as prospective parents or elderly people are not considered separately. Exposure to children is estimated for infants (<1 year) since this sub-group, compared with other children sub-groups, has a higher than expected exposure due to their high water consumption relative to bodyweight.

In estimating the exposure, the following assumptions are made:

- An infant of <1 year weighs 10 kg (US EPA, 2008) and an adult weighs 70 kg (enHealth, 2003);
- The drinking water consumed is 1 L/day for infants and 2 L/day for adults (enHealth, 2003);
- The controlled and reasonable worst-case lanthanum concentrations in drinking water are 2 and 33 µg/L, respectively; and

For both children and adults, the received lanthanum dose from oral exposure was calculated from equation 1:

Equation 1

$$D_{oral} = \frac{C_{ingested} \cdot V_{ingested}}{BW}$$

Where:

D_{oral} = Received dose via the oral route, µg/kg bw/day
$C_{ingested}$ = Concentration of lanthanum in drinking water, µg/L
$V_{ingested}$ = Volume of drinking water ingested, L
BW = Bodyweight, kg bw

The estimated received doses from direct oral ingestion of lanthanum for the controlled and reasonable worst-case exposures are shown in Table 5.2.

Table 5.2. Calculated daily received doses from oral exposure to direct ingestion of drinking water containing soluble lanthanum.

	D_{oral} (µg/kg bw/day)	
	Controlled	**Reasonable worst-case**
Child	0.2	3.3
Adult	0.06	0.94

The ingested amounts to which the majority of people will be exposed will be variable and possibly lower than these worst case estimates because water treatment processes used to prepare water as supplied to the consumer are expected to remove a proportion of the lanthanum ions (Section 9.4).

5.2.3 Recreational water

Public contact with the chemical may occur where the chemical has been spread over publicly accessible, recreational water bodies. The applicant intends that the public would be excluded from the aquatic area during the period of application and sedimentation. Following this, the mixing of the chemical with naturally occurring sediments should result in its presence at low residual concentrations only and therefore the risk of exposure of the general public to the notified chemical, and associated insoluble forms of released lanthanum, is considered low. However, people engaged in recreational activities both on and in the water may be exposed to the soluble lanthanum released from the product.

Oral exposure

For both adults (70 kg) and children (15 kg), it is assumed that inadvertent ingestion of water during swimming results in an approximate 500-fold lower exposure than drinking water. This conclusion is based on the estimated ingestion of 100 mL per session, 1 swim/day and 14 sessions per year (NHMRC, 2008).

Taking this into account, inadvertent ingestion of the soluble lanthanum present would result in a negligible internal oral dose.

Dermal Exposure

It is generally accepted that both soluble and insoluble forms of lanthanum cannot cross the intact stratum corneum (Shaklai and Tavassoli, 1982). In vivo studies in guinea pigs have shown that absorption of lanthanides through the skin is negligible (Inaba and Yasumoto, 1979 cited in Hirano and Suzuki, 1996) and, in vitro, the stratum corneum of intact rat skin is impermeable to water-soluble lanthanum nitrate (Jiang and Zhou, 2003).

It is therefore expected that during swimming, showering or bathing, dermal exposure of humans to the lanthanum present would result in a negligible internal dose.

6. Environmental Exposure

6.1 Sources of environmental exposure

6.1.1 Release from manufacture and import

At the time of the original assessment in 2001, the notifier indicated that up to 5000 tonnes per annum of Phoslock™ would be manufactured in Australia in the form of the aqueous slurry product at up to 30% solid (w/w) and it was assumed that all of this amount would be released into water bodies (5000 tonnes per year) in the form of the slurry based on projected annual manufacturing volumes in Australia after 5 years (NICNAS, 2001).

Since Phoslock™ slurry is no longer manufactured in Australia, with the chemical now imported as Phoslock™ granules from a manufacturing site in China in the form of the solid end-use product (PWS, 2007a and PWS, 2008c), no releases to the environment in Australia will occur as a result of the manufacture of this chemical or the formulated end-use product. Any accidental spills of Phoslock™ granules occurring during importation, storage, and transport are expected to be physically contained and swept up for reuse or for disposal to landfill.

The notifier stated that 317 tonnes of Phoslock™ granules were imported in the two years from 2007 to 2009 with no further imports anticipated (PWS, 2010). In a more recent data provided by the notifier, a further 144.9 tonnes were imported in 2011 (PWS, 2011b). This gives a total import volume of 461.9 tonnes Phoslock™ granules. Assuming the granules contain the average level of 5% lanthanum, the Phoslock™ granules would contain 23 tonnes of total lanthanum.

6.1.2 Release from applications to environmental waters

Phoslock™ is used exclusively to treat environmental water bodies which are susceptible to blooms of cyanobacteria by removing soluble phosphate from the water column and bind FRP released from sediments and from algal cells upon degradation. The specific use of Phoslock™ for water remediation applications could potentially result in releases to large and small water bodies including ponds, streams, rivers, lakes, and reservoirs that could potentially be affected by nuisance algal blooms. Thus, the sources of environmental exposure will mainly come from direct releases into these environmental waters. These releases may be contained in applications to ponds, lakes, and reservoirs, but releases may also occur in moving water bodies such as creeks and rivers, wherein the clay particles and dissolved and suspended lanthanum from the chemical may move from the initial application site. The fraction of total lanthanum in Phoslock™ that is readily released in environmental waters is a key factor in determining the potential aquatic hazard of the chemical. Some of the deliberate applications of Phoslock™ in various environmental waters in Australia and overseas are presented in Table 6.1. Further details of these applications are discussed in Sections 6.2.5 and 6.2.6.

Table 6.1. Phoslock™ (in granular or slurry formulation) treatment of environmental waters in Australia and overseas.

Location (Country)	Amount Applied (tonnes)
Deep Creek Reservoir (Australia)	55
Vasse River (Australia)	40
Canning River (Australia)	60
Torrens Lake (Australia)	62
Gnowangerup Dam No. 2 (Australia)	0.3
Lake Okareka (New Zealand)	60
Silbersee (Germany)	25.5
Bärensee (Germany)	11.5
Otterstedtersee (Germany)	11
Behlendorfsee (Germany)	214
Blankensee (Germany)	66
Eichbaumsee (Germany)	148
Het Groene Eiland (Netherlands)	14
De Rauwbraken (Netherlands)	20
Zwemplas de Kuil (Netherlands)	41.5
Clatto Reservoir (Scotland)	24
Loch Flemington (Scotland)	25
Hartbeespoort Dam (South Africa)	6

6.1.3 Release from disposal

The preparation of slurries of Phoslock™ at the application site from the granulated product is expected to limit the quantities of the chemical that require disposal to those small quantities of product granules that remain in the 25 kg import bags after use and residues on the application equipment. The wastage from unused solid product is expected to be ≤1% of the imported quantity of Phoslock™ (≤3.8 tonnes) based on typical wastage figures for solid products in bags. This quantity of chemical will be disposed of to landfills distributed across Australia together with the used import bags. The application equipment is expected to be cleaned on-site by flushing with water and the rinsates from this process are expected to be discharged into the previously treated water body. The relative quantities of residual Phoslock™ disposed of by this route are not expected to be significant even for relatively large water bodies given that the formation of slurries of Phoslock™ granules on-site is a continuous process. In this situation, the application equipment is effectively continually purged as each new bag of granules is mixed with water and applied to the water body.

6.2 Fate and transformation processes

6.2.1 Influence of physical and chemical properties

Phoslock Water Solutions (PWS) stated that the quantities of Phoslock™ applied in environmental waters will depend on a variety of factors including the surface area, volume and chemistry of the water body (PWS, 2008d). In the case of Phoslock™ slurry, a default surface application rate of 20 tonnes per hectare was assumed for the original environmental risk assessment (NICNAS, 2001). For Phoslock™ granules, the standard application rate for smaller water bodies is 25 kg/100 m^2 or 2.5 tonnes per hectare (PWS, 2008e). For larger water bodies, the company recommends application rates based on the mass of bioavailable filterable reactive phosphorus (FRP), mainly orthophosphate.

A dosing ratio of 100:1 for the mass of Phoslock™ granules to the mass of FRP is currently recommended (PWS, 2008d). The earlier laboratory trials employed a 200:1 dosing ratio to reduce the level of FRP from 1 mg/L to below detection limits (Ecowise, 2004 and 2005). The notifier stated that binding time for the 100:1 ratio is longer than the 200:1 ratio, however, they recommend the 100:1 ratio based on the maximum adsorption capacity of Phoslock™ to FRP at this dosing (PWS, 2011c). The 100:1 Phoslock™ to FRP ratio is equivalent to 5:1 on a mass basis of La to FRP (3.4:1 on a molar basis).

The effects of pH changes on Phoslock™ properties were examined by Ross et al. (2008). Water column samples were prepared by adding monopotassium phosphate (KH_2PO_4) to reproduce a 1 mg/L phosphorus level to reverse osmosis (RO) water, modifying the conductivity of the solution to 0.3 mS/cm using NaCl, and mixing the solution overnight at 200 rpm. The pH of the samples was adjusted to 5, 7, 8, and 9 utilising solutions of 0.1 M HCl and NaOH. For all the water column solutions, a Phoslock™ to phosphorus ratio of 230:1 was employed, wherein the granules were made into a slurry and added to the samples. Effects on Phoslock™ performance were monitored for 6 hours. After the sampling period, the phosphorus adsorption capacity was the same within the pH range 5-7 and decreased from pH 7-9 possibly due to the binding of ionic lanthanum to hydroxyl ions (Ross et al., 2008).

In the various Phoslock™ formulations, lanthanum is assumed to have replaced intercalated sodium and calcium cations in a bentonite clay host mineral phase. It is expected that the release of lanthanum from Phoslock™ preparations would be dependent on the following factors:

- the method of preparation wherein formulations developed by CSIRO minimised release rates over the more conventional preparation methods;
- the chemistry of the water being treated, namely phosphorus level, dissolved organic carbon (DOC), hardness, alkalinity, pH, and salinity; and
- the area and depth of the water body being treated (period of contact with water), dispersion, and degree of turbulence (mixing).

6.2.2 Environmental mobility and distribution

Where significant water currents exist, Phoslock™ is expected to remain in the water column and be transported. In water bodies with weak hydrodynamics, the clay particles comprising Phoslock™ will settle to the sediment. The bioavailability of lanthanum from Phoslock™ is principally due to the presence of the free lanthanum ions La^{3+} which are available for uptake by aquatic organisms.

Phoslock™ is intended to sequester phosphate anions as the constituent clay particles settle through the water column, and also to bind phosphate that may be released from the sediment compartment (PWS, 2006 and 2008d). The absorbed phosphate is expected to remain bound to Phoslock™ active sites ($\equiv La^{3+}$ active sites) (NICNAS, 2001 and PWS, 2006).

While Phoslock™ efficiently removes phosphate from waters with low alkalinity (<10 mg/L $CaCO_3$), it is less successful for phosphate removal in waters with higher alkalinity (>50 mg/L $CaCO_3$) because carbonate ion ($HCO_3^- \leftrightarrow CO_3^{2-} + H^+$, $pK_a \sim 10.4$) will compete with phosphate ($H_2PO_4^- \leftrightarrow HPO_4^{2-} + H^+$, $pK_a \sim 7$) for binding at Phoslock™ active sites ($\equiv La{\sim}PO_4 \leftrightarrow \equiv La{\sim}CO_3$). The phosphate and carbonate ions present in environmental waters would compete for the lanthanum active sites for binding and the resulting insoluble compound is not bioavailable. Dissolved water-column cations may also compete with lanthanum for binding with phosphate.

The lanthanum ion, La^{3+} in surface freshwaters could be expected to undergo hydrolysis reactions to form hydroxy species, react with phosphate and carbonate ions, and dissolved organics, and show a high propensity for adsorption to colloidal and suspended particles with negative surface charge. This expectation is supported by modelling of the speciation of lanthanides with a chemical speciation modelling program (MINEQL+ - Chemical Equilibrium Modeling System), which indicates that lanthanum complexes with carbonate and dissolved organic matter predominate in the surface waters of the Rhine-Meuse estuary in Netherlands. This study also concluded that pH, ionic strength, and alkalinity all significantly influence lanthanide speciation (Moermond et al., 2001).

A preliminary investigation of the speciation of the dissolved lanthanum fraction derived from Phoslock™ has been conducted in jar tests without added sediment in synthetic laboratory waters of varying alkalinities. The concentrations of lanthanum in the supernatant aqueous phase of mixtures of 20 g of Phoslock™ (presumably the granular formulation) tumbled for up to 96 hours in 200 mL of synthetic laboratory waters were determined by ICP-MS after passage through either a 0.45 µm filter or a 0.20 µm filter. The use of two different filter sizes was intended to indicate the presence of relatively high molecular weight lanthanum compounds in water samples filtered through a 0.45 µm filter, which is the first-pass operational definition for dissolved metal concentration in environmental water samples (ANZECC/ARMCANZ, 2000a). Relatively high molecular weight species such as colloidal lanthanum salts, which may be present in the sub 0.45 µm filter fraction of Phoslock™ treated environmental waters, are expected to have reduced bioavailability as compared with low molecular weight ionic lanthanum species, such as La^{3+}_{aq}, $LaOH^{2+}$, and $LaCO_3^-$ (PWS, 2008f).

The concentrations of lanthanum measured in filtrates from the 0.20 µm filters were, in most cases, lower than those measured for filtrates from the 0.45 µm filters. However, the precision of the analytical methodology employed was not reported and corrections for experimental artefacts such as adsorption of lanthanum ions on the filters employed were not provided. Consequently, it is not possible to draw firm conclusions about the significance of the apparent differences in measured lanthanum concentrations in filtrates passed through the two different filters. A series of new studies of the release of lanthanum from Phoslock™ have been conducted since the original assessment of this chemical. The most relevant of these are for the granular formulation of Phoslock™ in environmental water samples, and details are discussed in the succeeding sections.

6.2.3 Persistence and bioaccumulation

Phoslock™ is a fully inorganic particulate solid that is expected to accumulate in the sediment compartment of water bodies to which it is applied. The major component of Phoslock™ is bentonite clay which is typically composed of the smectite clay minerals, montmorillonite and beidellite (Deer et al., 1966), both of which are naturally occurring layered silicate minerals that are persistent in the environment on geological timescales. The primary clay particles of Phoslock™ that accumulate in the sediment compartment of treated water bodies will therefore persist indefinitely on biological timescales and enter the normal geochemical cycle for clay minerals. The lanthanum component of Phoslock™ will also persist indefinitely in the environment based on the natural abundance of the stable isotope of this element, ^{139}La, 99.91% naturally abundant (Audi et al., 2003).

Bioconcentration of Phoslock™ particles in aquatic organisms is unlikely since the size of dispersed particles is typically >1 µm and they are therefore not expected to cross intact biological membranes. However, there is the potential for colloidal clay particles and insoluble lanthanum colloids derived from Phoslock™ to bioaccumulate in filter feeders such as mussels (Hawker, 1990). This aspect of the environmental behaviour of Phoslock™

may require investigation given the likely exposure of filter feeders to elevated levels of lanthanum, and the high bioaccumulation potential for this element in aquatic organisms.

The bioaccumulation of lanthanum has not been well studied, however the limited available data indicate that there is a high bioaccumulation potential for lanthanum in aquatic organisms. The available data on bioaccumulation of rare earth elements including lanthanum up to June 2000 were reviewed by Netherlands' National Institute of Public Health and the Environment (Sneller et al., 2000). The data for lanthanum bioconcentration factors from this report are reproduced in Table 6.2.

Table 6.2. Bioconcentration factors (BCF) for lanthanum (Sneller et al., 2000).

Species/group	Freshwater (F) or Marine (M)	BCF (L/kg)
Amphipod	F/M	28840
Bivalves	F/M	15000-50000*
Worms	F/M	8000-120000*
Crustaceans	F/M	10000-40000*
Carp, muscle	F	0.22-1.10*
Carp, skeleton	F	3.66-8.11*
Carp, gills	F	11.2-18.8*
Carp, internal organs	F	634-978*

* These results are presented as ranges as the values for a series of lanthanide elements (including lanthanum) were reported together.

This review found some contradictory results for trends in the bioconcentration of some rare earth elements in aquatic biota. However, for lanthanum, the pattern is more consistent. Significant bioconcentration of lanthanum occurs in freshwater/marine amphipods (28840 L/kg (dry weight); unspecified species), freshwater/marine crustaceans (10000-40000 L/kg; unspecified species), freshwater/marine worms (8000-120000 L/kg; unspecified species), and freshwater/marine bivalves (15000-50000 L/kg; unspecified species).

A study in carp (*Cyprinus carpio*) revealed levels of rare earths including lanthanum in internal organs with a BCF of 634-978 L/kg (wet weight). These levels were substantially greater than the values measured for other organs (e.g. 0.22-1.10 L/kg (wet weight) in muscle). The largest measured bioconcentration factors quoted for rare earth elements are for water plants, where values up to 10^6 L/kg are given (Sneller et al., 2000). The exact location where the accumulated lanthanum is found within the organisms at a subcellular level has not been determined.

6.2.4 Fate of Phoslock™ in laboratory trials

De-ionised Water

In the original assessment for this chemical, the fraction of lanthanum released from Phoslock™ was determined to be <0.02% of total lanthanum based on the results of leaching tests in de-ionised water carried out at a solids-to-liquid ratio of 50 g of Phoslock™ in 1 L of water (NICNAS, 2001). From this, the fraction of total lanthanum is approximately <0.01 mg of lanthanum released per gram of the modified clay.

A more recent study has shown that the fraction of lanthanum released from a manufacturing slurry of Phoslock™ into reverse osmosis treated water at pH 6-7 is 1.25 mg of lanthanum per gram of solid, or >125 times the quantity originally estimated. By contrast, the fraction of readily released lanthanum from Phoslock™ granules in the same reverse osmosis treated water is 0.011 mg per gram of granules (IMT, 2004). Assuming that the batch of Phoslock™ granules used contained 5% w/w lanthanum, the quantity of lanthanum

readily released from Phoslock™ granules in de-ionised water is equivalent to approximately <0.022% of total lanthanum. This release rate is comparable to the originally estimated lanthanum release rate for Phoslock™ in de-ionised water (NICNAS, 2001).

The extent to which lanthanum is released from Phoslock™ will depend in part on the chemistry of the water phase to which it is applied. A series of laboratory and field tests on Phoslock™ granules discussed in later sections shows that the release of lanthanum from this chemical is dependent on variable water chemistry properties such as pH, alkalinity, hardness, and salinity. The availability of exchangeable cations, particularly higher charge cations such as Ca^{2+}, is expected to have a significant effect on release of La^{3+} via exchange processes. The fractional release of lanthanum from Phoslock™ in environmental waters varies significantly from the values measured in de-ionised water, as discussed in the succeeding sections. Hence, lanthanum release tests conducted in de-ionised water are not considered useful predictors of the concentrations of lanthanum in water that will occur as result of the use of Phoslock™ in environmental waters under realistic exposure scenarios.

Synthetic and Environmental Waters

The release of lanthanum from granular forms of Phoslock™ mimicking more realistic exposure conditions was tested in the laboratory using a batch of granules manufactured by Integrated Mineral Technology Pty Ltd (IMT) in China. The study was conducted in synthetic waters of varying salinity and in environmental water samples taken from a variety of field sites in Australia (Ecowise, 2005).

Although the precise geographical locations for the 14 sampling sites in QLD, NSW, ACT and SA were not presented in the study report, the type of water body sampled and some water chemistry parameters for the samples from each site (i.e. pH, EC, and FRP) were recorded. The location codes for the sampling sites, the type and initial water chemistry properties for each water body, and the treatment rate of Phoslock™ granules used in the laboratory-scale lanthanum release experiments are summarised in Table 6.3.

Table 6.3. Sample identification details, initial water chemistry characteristics, and dosing rates for environmental water samples used in the Ecowise (2005) study.

Water body (Sample ID)	Type (pH; EC, µs/cm; FRP, mg/L)	Phoslock™ (mg/L)
Bendora Reservoir, ACT (BEN)	Water supply reservoir (7.72; 26.5; <0.01)	50
Brisbane River, QLD (BRA)	River upstream (6.68; 2690; 0.146)	150
Brisbane River, QLD (BRB)	River estuarine (6.57; 21700; 0.21)	100
Brisbane River, QLD (BRC)	River estuarine (6.77; 35000; 0.07)	100
Brisbane River (duplicate), QLD (DUP)	River estuarine (6.77; 35000; 0.07)	100
Wivenhoe Dam, QLD (CCA)	Water supply reservoir (7.88; 333; <0.01)	100
Wivenhoe Dam, QLD (CCB)	Water supply reservoir (8.12; 267; <0.01)	50
Poets Corner Dam, NSW (PCA)	Irrigation dam (7.55; 2800; 0.01)	50
Maclean Lagoon, NSW (MAC)	Lagoon of a STP (6.83; 474; 9.78)	2000
Happy Valley, SA (SAW)	Potable water reservoir (8.40; 520; <0.01)	50
Homebush Bay, NSW (HOM)	Non-potable water (7.91; 1380; 0.743)	250
Queanbeyan,NSW (QUE)	Lagoon of a STP (7.9; 700; 0.012)	50
Penrith Lakes, NSW (PEN)	Recreational lake (8.07; 864; <0.01)	50
Moore Reserve, NSW (MOO)	Lake (8.14; 245; <0.01)	50
Midgee Research Group* (MID)	Lake (9.06; 1.68; <0.01)	50

* The location of this sampling site within Australia was not specified in the report.

The experiments were conducted at a single dose rate of approximately 200:1 Phoslock™ granules to phosphorus (based on the artificially adjusted FRP at time 0 minutes). Testing was conducted by the addition of appropriate amounts of the granules to 2 L volumes of sample water in beakers. For example, 2 g of Phoslock™ granules was added to 2 L volumes of synthetic laboratory waters with FRP levels artificially adjusted to 5 mg/L. The solid granules were dispersed in the water phase by mechanical agitation for 30 seconds with a glass rod, and sampling was performed at intervals up to 96 hours. A minisonde instrument was used for *in situ* measurement of pH, and an Eijklamp 3 in 1 probe for the electrical conductivity (EC) reading. The temperature was controlled (air conditioning) at approximately 20°C temperature. The salinity of synthetic laboratory waters was adjusted by the addition of NaCl or KCl.

The analysis of dissolved lanthanum was performed by ALS Laboratory Group with confirmation of random duplicates by Queensland Health Laboratory. The analytical techniques employed in the two laboratories were not specified, but are presumed to be either ICP-MS or ICP-AES. Filter syringes were used for water sampling and the water was acidified using HNO_3. Although few details were given regarding the filtration and acidification of samples taken for dissolved lanthanum analyses, the standard procedure is to pass the water through a 0.45 µm filter and acidify (by adding nitric acid, HNO_3) to pH <2 (ALS, 2006). Duplicate water samples from some tests were analysed for dissolved lanthanum in the two laboratories. The average relative percent deviation (RPD) between these duplicate measurements was 15.3%, with a maximum value of 39%. Other method validation parameters including accuracy and precision were determined as part of the study protocol.

The dissolved lanthanum concentrations in the supernatant aqueous phases of each mixture were measured at 1, 2, 6, 24, 48, 72, and 96 hours after initial mixing with Phoslock™ granules. The results from each test mixture are summarised in Figures 6.1 and 6.2, which were taken from Ecowise (2005).

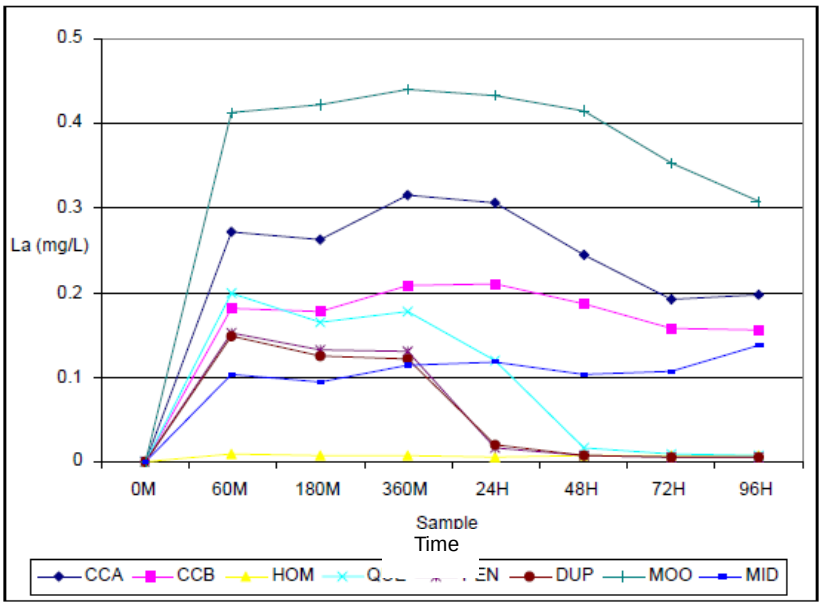

Figure 6.1. Dissolved lanthanum concentrations in a subset of environmental water samples treated with Phoslock™. Sample ID codes are defined in Table 6.3.

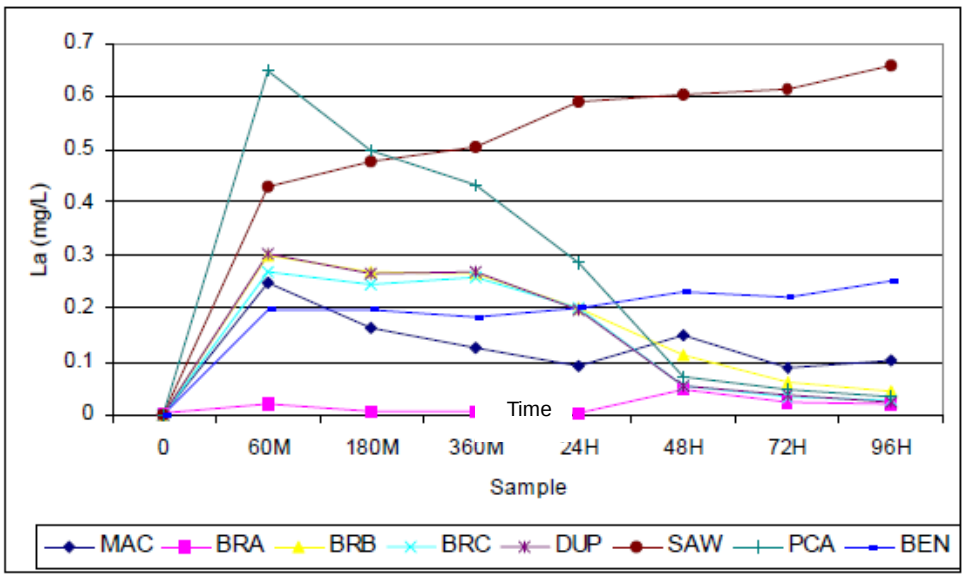

Figure 6.2. Dissolved lanthanum concentrations in a subset of environmental water samples treated with Phoslock™. Sample ID codes are defined in Table 6.3.

The release of dissolved lanthanum from Phoslock™ in diverse environmental waters reached peak or limiting values within 1 hour of mixing in most cases. The highest concentration reached after 1 hour was for the mixture with water from the irrigation dam at Poets Corner Vineyard Dam A (NSW) (Sample ID: PCA), where the dissolved lanthanum concentration was 0.649 mg/L. The dissolved lanthanum concentrations in half of the samples had declined or remained below 0.1 mg/L after 96 hours (e.g. samples from the Brisbane River: BRA-C). The exception to this trend was the sample from the Happy Valley potable water reservoir in SA (Sample ID: SAW). For this water sample, dissolved lanthanum concentrations increased monotonically over the course of the experiment up to the final value of 0.658 mg/L measured at 96 hours. A possible explanation for this

deviation is the presence of dissolved organic matter, which can react with lanthanum and hold it dissolved as irreversibly formed complexes.

Lanthanum release data from toxicity test reports

Levels of dissolved lanthanum for nominal doses of applied Phoslock™ were measured in toxicity tests conducted by Ecotox Services Australia (ESA, 2008), Clearwater (2004), and Clearwater & Hickey (2004). The toxicity effects are described in Section 8.2.

The cladoceran acute toxicity test by ESA (2008) used Phoslock™ slurry added to test waters with mixing by inversion at the start of the test. The samples were not stirred during the tests. It is uncertain whether inversion mixing enhances release of lanthanum over what might occur in a naturally settling Phoslock™ application. The chronic cladoceran test had daily solution renewals that maintained a more steady dissolved concentration. The nominal doses and dissolved lanthanum levels measured in these tests are shown in Table 6.4 and analysed further (Section 8.3).

Table 6.4. Lanthanum release from Phoslock™ granules during toxicity tests.

Test (Reference)	Nominal Phoslock™ dose applied, mg/L	Dissolved lanthanum levels, µg/L
Cladoceran, 2d	0	<1
(ESA, 2008)	0.25	3.4
	0.5	8.6
	0.75	11, 15[a,b]
	1.0	26, 17[a,b]
	2.0	48, 28[a,b]
	20	310, 210[a,b]
	50	480, 330[a,b]
Cladoceran, 7d	0	<1[b,c]
(ESA, 2008)	0.25	5.3[b,c]
	0.5	11.5[b,c]
	0.75	16.1[b,c]
	1.0	22.3[b,c]
Midge larvae, 38d	400	7.7
Midge larvae, 10d	400	140[d]
Amphipod, 10d	20	6.4[c]
	40	7.8[d]
	1000	483[d]
Rainbow trout	0	-
	20	8.2[d]
	40	10.1[d]
	200	13.6[d]
	400	-

[a] At 0h, 48h; [b] Dissolved lanthanum LOD & LOQ were 0.0022 & 0.02657 µg/L, respectively (Envirolab, 2009); [c] Mean of daily measurements; [d] Measured at the end of the test only.

In the case of Clearwater (2004) and Clearwater & Hickey (2004) studies, Phoslock™ was added as a 20 mL slurry to the surface of jars containing ~290 mL of water overlying ~210 g wet weight of a 70% sand and 30% mud sediment mix with the samples gently aerated. There was a clear difference in lanthanum release in these samples, with dissolved

lanthanum being in some cases a factor of 40-50 lower than in the cladoceran tests. It is possible that the initial release during sedimentation ceases once the Phoslock™ has settled to the sediment. Over the 10-day period, any chemical reactions of the released lanthanum are such that it is largely not in a dissolved form, although some of the settling particulate lanthanum may well be resuspended by the gentle aeration in the jar tests. Release of lanthanum into porewaters will result in binding to any soluble phosphate there, with lanthanum efflux into overlying waters likely to be slow. Given that lanthanum is supposedly trapped in the clay lattice, it is more likely that any soluble phosphate will react with the immobilised lanthanum so that porewater concentrations may indeed be low.

The jar tests on lanthanum leaching reported by Phoslock Water Solutions (PWS, 2007c), described in Section 6.2.5, indicate similar findings to the ESA (2008) test in that the concentrations remained high for 72 hours when water only suspensions of Phoslock™ were mixed (method not specified) and allowed to settle.

Salinity effects on lanthanum release

The efficiency of removal of 1 mg/L or 5 mg/L added FRP by Phoslock™ in synthetic laboratory waters of varying salinities was examined. The salinity range tested corresponds to values expected for freshwater (0 ppt halide salt) and brackish water (5 or 30 ppt halide salt). The dissolved lanthanum concentrations in the supernatant aqueous phase were measured at 1, 2, 6, 24, 48, 72, and 96 hours after mixing as part of this trial. The results for the dissolved lanthanum analyses are summarised in Figure 6.3, which is taken from Ecowise (2005).

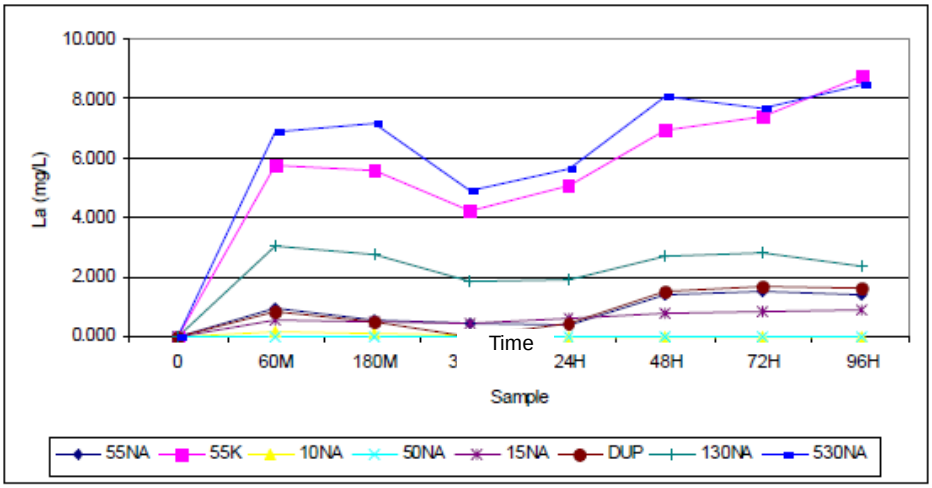

Figure 6.3. Dissolved lanthanum concentrations in synthetic laboratory waters treated with Phoslock™ granules under conditions of varying salinity and FRP.

Key: 55NA = 5 ppt NaCl (salinity) and 5 mg/L FRP; 55K = 5 ppt KCl and 5 mg/L FRP; 10NA = 0 ppt NaCl and 1 mg/L FRP; 50NA = 0 ppt NaCl and 5 mg/L FRP; 15NA = 5 ppt NaCl and 1 mg/L FRP.

Figure 6.3 showed that levels of dissolved lanthanum in the supernatant aqueous phase increased sharply from the initial measured value (<0.005 mg/L) to at least 0.157 mg/L 60 minutes after Phoslock™ granules were mixed with the test waters in all but one test mixture. The peak dissolved lanthanum concentration after 60 minutes of equilibration was 6.88 mg/L, which was measured in the 5 mg/L FRP + 30 ppt salinity (NaCl) test mixture. The highest measured dissolved lanthanum concentration was 9.03 mg/L in a duplicate sample taken from the same test mixture at the 96-hour time point. The only test that did not show a significant initial increase in dissolved lanthanum concentration was the 5 mg/L

FRP + 0 ppt salinity mixture, where the dissolved lanthanum concentration remained less than 0.005 mg/L over the course of the test.

The data presented in Figure 6.3 also revealed a strong influence of salinity on the levels of dissolved lanthanum released from Phoslock™ at a constant (initial) FRP. For example, the concentration of dissolved lanthanum after 96 hours in the test mixture with 30 ppt salinity (NaCl) and 5 mg/L FRP (9.03 mg/L La) was 5.4 times the level recorded for a mixture with 5 ppt salinity (NaCl) and 5 mg/L FRP (1.68 mg/L La) at the same time point, and >9030 times the level recorded in a mixture with 0 ppt added halide salt (<0.001 mg/L La). A similar trend was observed for dissolved lanthanum concentrations in test mixtures with initial FRP levels of 1 mg/L, although the respective peak concentrations of lanthanum reached in these samples were lower than for the corresponding test mixtures with 5 mg/L FRP.

The replacement of sodium by potassium in the added electrolyte also significantly increased the release of dissolved lanthanum from Phoslock™. The test revealed that after 96 hours, the concentration of dissolved lanthanum in a 5 ppt salinity test mixture adjusted with KCl (8.78 mg/L La) was 5.97 times the level for a mixture of the same nominal salinity adjusted with NaCl (1.47 mg/L La). The (initial) FRP in both mixtures was the same (5 mg/L) (Ecowise, 2005).

6.2.5 Fate of Phoslock™ in Australian field applications

Deep Creek Reservoir, New South Wales

The release of lanthanum from Phoslock™ granules in environmental water samples taken from the Deep Creek Reservoir, a water body that has an ongoing problem with blooms of cyanobacteria, in the Eurobodalla Shire of southern NSW was assessed in laboratory scale jar tests by Phoslock Water Solutions (PWS) prior to application of the chemical to this water body. Although not explicitly stated, it is presumed that the sample of Phoslock™ granules used in these tests was the so-called "Eureka 1" formulation that was subsequently used to treat the reservoir. These tests also involved an investigation of the ability of added soda ash (Na_2CO_3) to increase the alkalinity of the water and mitigate the release of dissolved lanthanum under environmentally relevant conditions (PWS, 2007b).

A number of relevant water chemistry parameters for a sample of the reservoir water was measured as part of the laboratory scale tests (PWS, 2007b). The parameters measured for this sample are summarised in Table 6.5. The date of sampling, the volume of water collected, and the sampling protocol were not reported. However, based on descriptions and the date of the report, it is inferred that >20 L was collected in, or shortly before, January 2007. Water hardness, the combined concentration of calcium cations (Ca^{2+}) and magnesium cations (Mg^{2+}) in the water sample as determined by ICP-AES, in a sample from the Deep Creek Reservoir used in this study is 27.6 mg/L $CaCO^3$. This hardness value is in the range usually defined for soft water (<60 mg/L as $CaCO_3$; Health Canada, 1979).

Table 6.5. Water chemistry parameters for a sample taken in or before January 2007 from Deep Creek Reservoir.

Water Chemistry Parameters	Value
Filterable Reactive Phosphate (ppm)	0.02-0.04
pH	7.19
Alkalinity (mg/L as $CaCO_3$)	24-29
La^{3+} (mg/L)	<0.003
Na^{+} (mg/L)	15
Ca^{2+} (mg/L)	4.3
Mg^{2+} (mg/L)	4.1
Si (mg/L)	5.4

The FRP level in the water sample was evaluated by the molybdenum blue colorimetric method. Although the units for FRP were not explicitly defined in the report, it is assumed that the reporting of these results conforms to conventional practice in which the concentration is expressed in terms of the equivalent mass of elemental phosphorus per volume. In this case, the range of concentrations reported for FRP is equivalent to 20-40 µg/L of elemental phosphorus, which is above the default trigger value of 5 µg/L P for phosphorus stressor levels in freshwater lakes and reservoirs in southeast Australia and the pH of the water is consistent with a water body in the middle of the acceptable range of 6.5-8.5 for low-land rivers in NSW in accordance with the *Australian and New Zealand guidelines for fresh and marine water quality* (ANZECC/ARMCANZ, 2000b). The DOC and ionic conductivity of the water sample were not reported.

The release of lanthanum from Phoslock™ granules in the reservoir sample was assessed by mixing small quantities (≤200 mg) of a sample of the granules from a batch manufactured in 2005 with 2 L volumes of reservoir water in jars. It is not clear from the report whether most of these test mixtures were mechanically mixed and, if so, for how long. Therefore, it is assumed that most solid samples of Phoslock™ granules were initially evenly mixed with reservoir water in the jars by mechanical means and then were allowed to stand for the remainder of the test. The sampling protocol consisted of acidifying and filtering 10 mL sub-samples of water taken from these jars at selected intervals following mixing. The type of filtration employed was not specified, but since the filtrates were taken for metal analysis by ICP-AES it is presumed that this filtration step involved a 0.45 µm filter. In this case, the metal analysis results of the filtered sub-samples are taken to indicate the levels of dissolved metals present in the supernatant water-phase in the test jars at various time points.

The concentration of dissolved lanthanum released into the supernatant water in the test jars was tested at three treatment rates of 12, 28 and 100 mg/L over a 72-hour test interval. The lowest treatment rate in this laboratory test is equivalent to the treatment rate subsequently used in the field application of Phoslock™ granules in the reservoir in April 2007 (PWS, 2007b). The dissolved lanthanum levels in the supernatant liquids of each test mixture was evaluated at time t = 0 (assumed to be shortly after mixing the reservoir water and Phoslock™), 24, and 72 hours. The increase in measured lanthanum levels between the 24- and 72-hour sampling was <15% in all supernatant liquid phases, which indicates that partitioning of lanthanum between the water and solid phase reached (or was near to establishing) equilibrium under the test conditions after 72 hours. The analysis for lanthanum in the water phase after 72 hours at the three treatment rates are summarised in Table 6.6. The deduced mass fraction of lanthanum initially present in Phoslock™ granules that is found in the water phase after 72 hours is also summarised.

Table 6.6. Lanthanum releases from Phoslock™ granules into Deep Creek Reservoir laboratory water samples after 72 hours at three different application rates.

Phoslock™ treatment (mg/L)	Dissolved lanthanum level after 72 hours (mg/L)	Total lanthanum in Phoslock™ granules released in the water (%)*
12	0.102	17
28	0.173	12.4
100	0.271	5.4

* Calculations assume 5% w/w lanthanum in Phoslock™ granules

The results from this trial revealed significant releases of lanthanum from the granules in a sample of environmental water. To illustrate, the quantity of lanthanum released into Deep Creek Reservoir at a field application rate of 11 kg of Phoslock™ granules per megalitre is equivalent to at least 17% of the total mass of lanthanum in a typical batch of this product. This level of lanthanum release is too large to be accounted for based simply on the dissolution of residual lanthanum salts co-precipitated with Phoslock™ during the manufacturing process. Instead, these high levels of lanthanum release appear to indicate release of intercalated lanthanum from the constituent clay particles of the chemical. It was argued that this may be due to the presence of silica in the environmental water sample from the reservoir, but no supporting evidence for this hypothesis other than analysis results for total silicon were provided (PWS, 2007b).

The results from these tests can only be taken as minimum release figures. The lanthanum that precipitated as insoluble phosphate and carbonate salts and the residual lanthanum remaining in the Phoslock™ clay particles were not measured. More definitive measures of the fractional release of lanthanum would ideally be met by comprehensive mass balance measurements including the proportion of lanthanum in all phases in dissolved metal, colloidal and particulate lanthanum salts, and the primary clay particles of Phoslock™. Although there are limitations to the predictive power of this leaching trial, it demonstrated that the apparent fractional release of total lanthanum decreases with increasing dose rate of Phoslock™ in an environmental water sample.

The field application of Phoslock™ in the Deep Creek Reservoir occurred in 2-5 April 2007 (NSW DECC 2008a & PWS, 2007c) wherein 55 tonnes of the "Eureka 1" granular formulation and 5 tonnes of soda ash (Na_2CO_3) were sprayed onto the surface waters from a small boat in the form of an aqueous slurry with the soda ash aiming to reduce the ionic lanthanum by 50% (PWS, 2007c). After this treatment, an extensive kill of small snails was observed on the shoreline and in the reeds of the reservoir on 5 April 2007 (NSW DECC, 2007b) and two days after, a number of small dead fish were observed (NSW DECC, 2007b). The death of fish apparently continued for a further two weeks following application. A biological survey of water samples taken from the reservoir about 7 weeks after application also found almost no living zooplankton, considered to be unusual for this type of water body (NSW DECC, 2007b), noting that pre-application monitoring was not carried out.

Interpretation of the results from the application at Deep Creek Reservoir is complicated by the use of soda ash, because this would be expected to significantly and rapidly change the pH of the reservoir. The extent to which ecological effects are associated with lanthanum compared with pH change cannot be definitively determined based on the available data.

This field application in Deep Creek Reservoir used approximately three times excess of Phoslock™ relative to FRP content of the whole water body. Field trials for which sufficient data are available to calculate the ratio of Phoslock™ to FRP are listed in Table 6.7, which

shows the application to Deep Creek Reservoir used a much larger excess of Phoslock™ than any of the other field applications reported here.

Baseline environmental chemistry parameters and ecotoxicological health indicators for the reservoir have not been conducted. Thus, the environmental health status of this water body before and after Phoslock™ treatment cannot be evaluated. However, it is known from measurements made by Phoslock Water Solutions and the then NSW DECC that the concentration of dissolved lanthanum in the waters of Deep Creek Reservoir increased by a factor of >70 from <3 µg/L before application (PWS, 2007b) up to 220 µg/L after application of Phoslock™ (NSW DECC, 2007b) with the dissolved lanthanum level still elevated (98 µg/L) at up to 4 weeks following application (NSW DECC, 2007a).

The peak concentration of dissolved lanthanum measured in the waters of the Deep Creek Reservoir was more than 10 times greater than the Predicted Environmental Concentration (PEC) for lanthanum (20 µg/L) that was calculated in the original environmental risk assessment (NICNAS, 2001). The much higher than expected release of lanthanum measured in the field application is consistent with the laboratory scale tests conducted by the company on environmental water samples taken from the reservoir which showed significant releases of contained lanthanum from Phoslock™ granules at field application rates.

In this particular case, the finding that significant levels of dissolved lanthanum persisted for long periods after application of Phoslock™ to an environmental water body also indicates that precipitation of lanthanum as insoluble phosphate and carbonate salts is not always sufficient to rapidly remove La^{3+} released from the chemical, especially in soft waters. This latter finding indicates that the chemistry of some environmental water bodies may not be sufficient to mitigate the environmental risks arising from the higher than expected releases of dissolved lanthanum from Phoslock™, as previously assumed.

Vasse and Canning Rivers, Western Australia

The Vasse and Canning Rivers are freshwater systems frequented by summer blue green algal blooms. The application and monitoring of Phoslock™ were conducted by the CSIRO and the WA Water and Rivers Commission. Surface and bottom waters were collected and analysed as part of the water quality monitoring regime for the rivers. In the Vasse River, initial treatment of 20 tonnes was conducted on October 2001 with succeeding applications of 10 tonnes each on December 2001 and January 2002. Water collection and analysis were performed for treated and control sites. The authors claimed that the FRP levels at the treated and control sites were ≤5 and ≤200 µg/L, respectively (data not presented). In the Canning River, initial treatment of 30 tonnes was conducted on November 2001 with succeeding applications of 15 tonnes each on January and February 2002. FRP levels for the treated sites in the Canning River were obtained and found to have similar results as the Vasse River with an observed decrease in pre-treatment FRP of 50 µg/L to 20 µg/L after application (Robb et al., 2003).

Torrens Lake, South Australia

Torrens Lake is an in-stream urban lake on the Torrens River in Adelaide that has been affected by persistent summer blooms of cyanobacteria (SA EPA, 2007). In response to a cyanobacterial bloom in February 2007, 50 tonnes of Phoslock™ granules were added to the lake in 6-8 March 2007. The chemical was added to the surface waters of both the western (30 tonnes) and eastern (20 tonnes) reaches of the lake from a small boat.

The water quality of Torrens Lake was characterised prior to the first application of the product which provided baseline data for monitoring the physical, chemical and biological effects of the addition of this chemical. The water quality properties that were measured included pH (8.0-8.4, initially), turbidity (20-50 NTU), conductivity (1450 EC) and total dissolved solids (800 mg/L), algal biomass (50-112 µg/L, based on chlorophyll

concentrations), dissolved oxygen (6-8 mg/L O_2), and oxidation-reduction potential (<20 mV, reference electrode not specified). The initial concentration of soluble phosphorus (FRP) was at or below the limit of detection (LOD) of the analytical method (0.005 mg/L) at all sampling sites. The measured levels of alkalinity and DOC in the lake were not reported (AWQC, 2007). However, the lake waters were described as both hard and high in dissolved organic carbon (SA EPA, 2007).

The levels of soluble and total lanthanum were measured at three mid-channel sites within the lake immediately before, and at weekly intervals up to 2 months after application. At each site, samples were taken from the surface, mid-depth, and bottom waters of the lake. Samples of surface water were also taken from the river bank near each mid-channel sampling site. In addition, measurements of both soluble and total lanthanum were made at five sites in the Torrens River downstream of the lake on the day before Phoslock™ was applied and on two separate days following significant rainfall events on 25 March and 28 April. The details of the sampling protocol and analytical methodology used for these measurements were not provided. The LOD for lanthanum was indicated to be 1 µg/L. The concentration of both total and soluble lanthanum prior to application of Phoslock™ to Torrens Lake was less than the LOD at all monitoring sites (AWQC, 2007). The soluble lanthanum concentration metric characterised in the chemical monitoring protocol was not explicitly defined but is assumed to conform to the usual operational definition of dissolved metals (i.e. the fraction of total metals that pass through a 0.45 µm filter).

The peak total lanthanum concentration in the lake was 2.43 mg/L was measured in surface waters near the bank at one site on the day of Phoslock™ application. The level of total lanthanum at all sampling sites declined relatively rapidly within the first 7 days after application to a plateau concentration of approximately 0.1 mg/L and persisted at this level for up to 7 weeks at some sites. The two significant rain events on 25 March 2007 and 28 April 2007 resulted in increased water flows through the lake, however, they did not appear to result in significant elevation of the levels of total lanthanum above the plateau concentration of approximately 0.1 mg/L (AWQC, 2007).

The peak soluble lanthanum concentration in the lake was 110 µg/L, which were measured in mid-channel surface waters on the day of Phoslock™ application. An initial peak in the concentration of soluble lanthanum was also found at all other sampling sites during the application of Phoslock™ (or on the day after) although it was <25 µg/L in each case. At most sampling sites, the measured concentration of soluble lanthanum decreased monotonically for the first 7 days after application. The concentrations were <20 µg/L at all sites, and below 10 µg/L at most sites, 7 days after application of Phoslock™ was complete. The levels of soluble lanthanum remained elevated above the initial background level (<1 µg/L) when the last measurement was taken on 1 May 2007.

A second peak in soluble lanthanum concentration was detected at all sampling sites after the rain event on 25 March 2007. The peak concentrations measured between 27 March and 10 April 2007 were all ≤24 µg/L. For most sampling sites, the levels of soluble lanthanum measured during the second peak were comparable to or greater than concentrations measured during the application of Phoslock™. The levels remained elevated for between 16 and 23 days compared with levels of soluble lanthanum measured before the rain event. The concentration at all sites was <5 µg/L by 24 April (30 days after rain), however, concentrations did increase again slightly to approximately 5 µg/L at some sites following a second rain event on 28 April 2007. The increased levels of soluble lanthanum in Torrens Lake after the first rain event do not appear to have been correlated with total lanthanum concentrations which remained at or near their plateau levels of approximately 0.1 mg/L over this sampling interval at most sites.

The Australian Water Quality Centre (AWQC) of the SA Water Corporation suggested that the increase in soluble lanthanum levels after a rain event may have occurred as the result of

resolubilisation of lanthanum from complexed material in the water column or sediments. This could be a result of an influx of organic matter entering the lake with the increased water flows, which was indirectly detected by a decrease in the dissolved oxygen concentration in lake waters to a minimum of 2 mg/L O_2 after the rain event. There were also other changes in the water chemistry such as a decline in pH from 8.1 to 7.5 that may have resulted in release of more soluble lanthanum species (AWQC, 2007).

The rainfall events were also correlated with an increase in the levels of total lanthanum measured at several sites downstream of the lake's weir. The peak total lanthanum concentration measured after the first rainfall event was 16 µg/L, well above the pre-application background levels of <1 µg/L, although less than the typical levels of total lanthanum in the lake at that time (approximately 100 µg/L). The total lanthanum levels downstream were also elevated after the second rainfall event (≤8 µg/L). The levels of soluble lanthanum were not as well correlated with the rainfall events as no soluble lanthanum was detectable before or after the first rainfall event. However, a small elevation in soluble lanthanum concentrations was measured following the second rainfall event (≤4 µg/L) (AWQC, 2007). Although the concentrations of total and dissolved lanthanum measured downstream of the lake's weir after the rain event were relatively low, they represented measurable increases above the background levels and provide evidence of off-site movement of lanthanum from an impounded water body.

The chemistry of the sediment compartment in Torrens Lake was investigated in a follow-up monitoring study carried out approximately 7 months after Phoslock™ was first applied in the lake. This study involved analysing total lanthanum, total phosphorus and FRP in 5 cm deep sediment core samples and the overlying water. The samples were taken from the bank and centre of the lake at three locations within the treated area of the lake. Sediments from an upstream sampling site were used as a negative control sample and a site downstream of the weir was also analysed. The LOD was 1 µg/L and the total lanthanum levels in the sediment samples were reported in mg/L (AWQC, 2008a).

The concentration of total lanthanum in all overlying water samples was ≤0.002 mg/L. This result indicates that the elevated levels of both total and dissolved lanthanum that remained present in the water compartment of Torrens Lake approximately 2 months after treatment with Phoslock™ had declined to near the pre-application background level (<1 µg/L) after a further 5 months.

The sediment from within the treated area of Torrens Lake revealed high concentrations of total lanthanum in most samples. The highest concentration measured was 721 mg/L in a single upper sediment layer sample from the centre of the lake. The concentrations at other sites were lower (≤286 mg/L). In all cases the concentrations were higher in sediments taken from the centre of the lake than the samples taken from the bank. It is not clear whether this reflected an artefact of the method by which Phoslock™ was applied or related to the internal hydrology of the lake (AWQC, 2008a).

The background levels of lanthanum in sediments prior to application of Phoslock™ were not provided making it difficult to make definitive conclusions regarding the extent to which addition of Phoslock™ changed the total quantities of lanthanum present in the lake's sediments. However, if the total lanthanum level in sediments below the lake's weir (<25 mg/L) is taken as representative natural background lanthanum, the total lanthanum levels in the top 2.5-cm layer of sediment within the treated area of the lake were elevated by a factor of >7- to >33-fold, 7 months after application of Phoslock™.

The upstream negative control site showed a relatively high total lanthanum concentration of 126-137 mg/L that was comparable to some results from within the area of Torrens Lake treated with Phoslock™, and significantly above the levels measured in sediments below the weir. This unexpected result was attributed by the study authors to upstream transport of

Phoslock™ clay particles adsorbed to sediments (AWQC, 2008a). An alternative hypothesis is that there is significant variability in total lanthanum levels within the sediment of the Torrens River which may indicate that the elevation of total lanthanum found within the sediments of the treated area of Torrens Lake and at the upstream sampling site is not as significant as assumed.

The authors argued that the levels of total lanthanum measured in the sediment of the treated area of Torrens Lake and the generally lower concentrations of FRP in the water column above the treated sediments as compared with the upstream control provided evidence of the formation of a "barrier layer" to the flux of bioavailable phosphorus from the sediment to the overlying waters of the lake (AWQC, 2008a). However, the two metrics used provide only indirect evidence for the presence of Phoslock™ within the sediment. For example, measurements of total lanthanum do not distinguish contributions from lanthanum present in other phases such as lanthanum phosphate which is no longer available for binding more phosphate or the fraction adsorbed on particles or in colloidal form, or dissolved lanthanum in pore waters.

In a follow-up treatment, 12 tonnes of Phoslock™ were applied on 25 February 2008 to reduce the likelihood of cyanobacterial blooms in preparation for recreational events scheduled in the weeks following treatment. The water treatment and sampling methods were similar to the procedures of the first Phoslock™ application in Torrens Lake.

The water quality parameters measured three days prior to treatment were pH (8.4-10.2), turbidity (19-35 NTU), conductivity (1143-1530 EC), algal biomass (41.2-51.8 µg/L chlorophyll), dissolved oxygen (6-15 mg/L O_2 at the water surface), and oxidation-reduction potential (up to 190 mV, reference electrode not specified). The initial FRP level was usually below LOD (0.005 mg/L) at all sampling sites except for one site which had a level of 0.01 mg/L. Initial total and soluble lanthanum levels of <0.03 and <0.001 mg/L, respectively, were detected at the sites. It is assumed that the same method was applied in determination of lanthanum levels in the previous application.

The same parameters above were also measured as part of the intensive post-application monitoring for eight weeks (up to 23 April 2008) and then monthly monitoring up to September 2008. The methodology employed for sampling and analysis of the water quality parameters in the post-application monitoring stage are assumed to be similar to the monitoring of the previous application in the lake. There were some unreliable pH readings on 25-26 February and 2 March 2008 due to problems with the pH probes. The maximum total lanthanum levels detected on the day of Phoslock™ application were 1.12 mg/L measured on the bank of the treated site and 0.086 mg/L on the control site. The total lanthanum levels markedly decreased in the treated site in the three days after the product was applied. On the 6th day post-application, the total lanthanum levels were <0.08 mg/L at the treated site and remained below this value during monitoring, except for a spike on 18 March 2008, probably due to sediment resuspension (AWQC, 2008b).

Soluble lanthanum had maximum concentrations of 12 and 2 µg/L at the treated and control sites, respectively, on the day of application. By the 6th day after Phoslock™ treatment, the soluble lanthanum concentrations remained below 2 and below 1 µg/L at the treated and control sites, respectively. A slight increase (1 µg/L at the treated site) was detected after a rain event in late March 2008.

Dissolved oxygen decreased in the first week post-application at both the treated and control sites but recovered and remained stable for the duration of the monitoring. The authors stated that the temporary decrease in dissolved oxygen after the treatment could be caused by the rapid algal cell loss following flocculation and decomposition (AWQC, 2008b).

Gnowangerup Dam No. 2, Western Australia

The waters of the Gnowangerup Dam No. 2 in WA were treated with 300 kg of Phoslock™ (presumed to be the granular formulation) in a trial carried out on 30 January 2008. The application of Phoslock™ to this reservoir was achieved with a shore based applicator system (Total Eden system). The concentration of orthophosphate (as P) in the waters of this potable water reservoir was adjusted to 30 µg/L prior to the trial by the addition of orthophosphate solution, which was intended to replicate levels of bioavailable phosphorus associated with algal blooms. The water chemistry of the reservoir was not described except to note that the levels of organic bound and inorganic phosphorus were below the LOD prior to augmentation with orthophosphate. Detailed description of the trial and a complete set of monitoring data were not available for this assessment, but summary information was provided in the form of a preliminary report from the Water Corporation of WA (Water Corp. WA, 2008).

The levels of lanthanum in the water column and sediment compartment of the reservoir were determined. The protocol included measurements of total lanthanum and filterable lanthanum (0.45 µm filter) for samples taken from both compartments before and immediately after application of Phoslock™. The details of the sampling protocols and analytical methodology used to obtain these measurements were not presented in this preliminary report.

The maximum concentration of total lanthanum in the water column was in the range 200-250 µg/L at three sampling sites immediately after application of Phoslock™. The levels of total lanthanum at all sampling sites had declined to below 20 µg/L seven days after treatment. The dissolved lanthanum concentration apparently remained below 20 µg/L.

The levels of total lanthanum in the sediment compartment at one sampling site increased from a pre-application level of approximately 20 mg/kg (weight basis not specified) to a post-application peak of 55 mg/kg after 7 days. The level of filterable lanthanum in this compartment remained below the LOD (reported as 5 µg/L) (Water Corp. WA, 2008).

6.2.6 Fate of Phoslock™ in overseas applications

Phoslock™ (granular form) has been applied in several water bodies in various countries (PWS, 2011a). The application details and lanthanum releases from each overseas treatment are presented in this section.

New Zealand

The Lake Okareka remediation plan was developed by the Environment Bay of Plenty, the Rotorua District Council, and the Te Arawa Lakes Trust, including consultation with the lake's community, with the aim of reducing the nutrient load of the lake. The plan involved lake treatment to remove phosphorus by 100 kg/year for three years. Phoslock™ application had been nominated to achieve the projected reduction (McIntosh, 2007).

Fortnightly monitoring of the lake in 2004/05 showed that the pre-application phosphorus load from the lake's sediments was 130 kg/year for the whole lake. The Environment Bay of Plenty applied 60 tonnes of Phoslock™ granules (three times with 20 tonnes per application) on 15-17 August 2005, 27-29 June 2006, and 20-21 March 2007 from a rotary spreader mounted on a barge. The water quality was monitored for 26 months with measurements of the following parameters: pH, dissolved oxygen, temperature, total lanthanum, phosphorus, nitrogen, turbidity, and chlorophyll.

These measurements revealed that total lanthanum levels in the upper part of the water column peaked at approximately 120 µg/L immediately after application of Phoslock™. These measurements also revealed that the total lanthanum levels declined to below 20 µg/L relatively quickly (within 1 month) and to pre-application levels within 5 months. The

decline in the elevated levels of total lanthanum was presumably due, at least in part, to particles of Phoslock™ containing intercalated lanthanum settling through the water column onto the underlying sediment.

The dissolved lanthanum levels in the lake waters following Phoslock™ application were not reported. The extent to which the measurements of total lanthanum reflect the presence of lanthanum bound within the modified clay particles compared with dissolved or particulate lanthanum phases cannot be assessed. The author stated that phosphorus load in the sediment has been reduced by nearly 100 kg/year with a strengthened phosphorus absorbing capacity of Lake Okareka expected for 3-4 years (McIntosh, 2007).

Germany

Phoslock™, known as Bentophos® in Germany, had been applied in several lakes wherein all the field applications and monitoring were performed by the German company Bentophos GmbH. The applications of the product in all of the lakes were done to reduce the phosphorus concentrations brought about by sediment release. In addition, the lakes had several incidences of major blue green algal blooms.

The Silbersee in Stuhr is used for fishing and other recreational activities. The lake's sediments have been reported to have a high nutrient loading with previous instances of deep water removal to reduce the nutrients. The application of Phoslock™ was identified by the Stuhr community as a strategy to reduce phosphate from the lake (IDN, 2008a). Water quality monitoring was conducted in the lake before Phoslock™ application. The total phosphorus was measured wherein approximately 40% was bioavailable by Psenner fractionation. The company applied 21.5 tonnes of Phoslock™ granules on 14-15 November 2006 using a pontoon-based mixing system wherein the granules were mixed with the lake's waters then sprayed over the lake's surface. Post-application monitoring was conducted until August 2009. The levels of total lanthanum decreased from 100 μg/L following application to 4 μg/L at the end of the monitoring period (IDN, 2009a).

In the City of Bruchköbel is the Bärensee, a recreational lake with frequent occurrences of heavy blue green algal blooms. Phosphorus immobilisation through Phoslock™ application was identified by the City of Bruchköbel in consultation with the water authorities of the district of Main-Kinzig, the health authorities, and the Hessen Land Office for Environment and Geology (IDN, 2008b). Similar to the method employed in the Silbersee, 11.5 tonnes of Phoslock™ was applied in the Bärensee on 12-13 June 2007 to reduce the phosphorus load of the lake's sediments. The initial level of phosphorus and the product's recommended dosing were used to estimate the amount of Phoslock™ applied on the lake. Pre- and post-application monitoring (up to July 2009) of the lake's water quality was conducted. Total lanthanum levels were 130 μg/L following application and decreased to <10 μg/L at the end of the monitoring period (IDN, 2009b).

Netherlands

In Netherlands, Phoslock™ has been applied to at least three lakes. All applications and monitoring were conducted by one or more of the following organisations: Phoslock Europe GmbH, Institut Dr Nowak (IDN) and/or the Aquatic Ecology Department of Wageningen University.

Pre-application monitoring in Het Groene Eiland in 2006 and 2007 showed chlorophyll-a levels of up to 62 μg/L and variations in total phosphorus concentrations (0.10-0.92 mg/L). Using a pontoon-based application system, 11 tonnes of Phoslock™ were applied on 16-17 April 2008 with the aim of removing 110 kg phosphorus from the water column and the sediment of the lake. The lake's water quality was analysed monthly up to September 2008. Total lanthanum was analysed, employing the ISO 11885-E22:1997-11 method, with a pre-application level of <0.002 mg/L and remained below 0.03 mg/L after application. The

company reported a "natural algal crash" on 22 May 2008 possibly from ash runoff brought about by burning of logs in close proximity to the lake. In the lake's sediments, total lanthanum concentrations were as follows: 23, 24, and 200 mg/kg dw on 13 December 2007, 16 April 2006, and 22 May 2008, respectively (Yasseri, 2008). A second Phoslock™ application of 3 tonnes was conducted on 31 March 2009 on the deeper areas of the lake. The Secchi depth, phosphorus levels, chlorophyll and phaeopigment concentrations were analysed and presented in a graph. Lanthanum levels were not reported before and after this application (PWS, 2010).

De Rauwbraken is a recreational lake with incidences of cyanobacterial blooms. The company applied two tonnes Phoslock™, two tonnes polyaluminium chloride as flocculant buffered with 75 kg calcium hydroxide, and 16 tonnes of Phoslock™ over three successive days on 21-23 April 2008 using a pontoon-based application system. The mean total phosphorus concentration was significantly reduced from the pre-treatment level of 91 to 19 μg/L after application of the treatment combination. Lanthanum levels were 0.01 (total) and 0.01 (filtered) μg/L at pre-application and markedly higher post-application with levels of 253.1 (total) and 28.0 (filtered) μg/L (van Oosterhout & Lürling, 2011).

A treatment combination of iron chloride (40%, 3000 L, 4.38 tonnes) flocculent, calcium hydroxide (200 kg) pH buffer, and Phoslock™ (41.5 tonnes) was applied in the recreational lake Zwemplas de Kuil to reduce the lake's algal population. The aim of the addition of iron chloride and calcium hydroxide was to transport the algal biomass to the sediment to prepare for Phoslock™ treatment. The flocculent and pH buffer were applied on 18 May 2009 in a pontoon-based system. Phoslock™ was administered on 19-21 May 2009, with the first 13.65 tonnes spread over directly to the water surface and the remaining 27.85 tonnes applied through hypolimnetic injection at a depth of 5 m. Pre-application sampling and analysis of the lake's water quality was done once in 18 March 2009 and post-application monitoring done four times up to 9 March 2010. After Phoslock™ application, total and filtered lanthanum concentrations were up to 0.01 and 0.009 mg/L, respectively (van Goethem, 2010).

Scotland

Field applications of Phoslock™ in Scotland were conducted to reduce the occurrence of cyanobacterial blooms from nutrient loading in the lakes. The treatment and monitoring were administered by the company concurrent with monitoring activities by the Scottish Environment Protection Agency (SEPA) and the Centre for Ecology and Hydrology (CEH) of Scotland's Natural Environment Research Council. Before any Phoslock™ application was made, the company applied for a controlled activity licence through SEPA. The licensing procedure involved preparing a risk assessment for Phoslock™ treatment, which included monitoring assessment plans and stakeholder consultation.

The Clatto Reservoir previously supplied water to the City of Dundee and is currently used for recreational and educational activities. Groves (2009) stated that the evaluation conducted by SEPA in August 2008 demonstrated diverse blue green algae species with high algal counts. Water quality testing was performed twice before Phoslock™ application and the level of phosphorus determined from the analysis was used to estimate the dose rate required to reduce the nutrients of the reservoir. Phoslock Europe GmbH obtained a SEPA controlled activity licence prior to applying 24 tonnes of Phoslock™ granules on 4-5 March 2009.

After application, reservoir water samples from 1-m and 4-m depths were collected monthly (up to August 2009) by peristaltic pump and sent to the German laboratories of the Institut Dr Nowak for water quality analysis. SEPA assessed the algal counts from water samples on 6 occasions (two occasions each month in June, July, and August 2009). At the 1 m depth, total lanthanum concentrations decreased from a peak of 0.21 mg/L on the day of

application to 0.03 mg/L on 19 March 2009 to <0.01 mg/L at the end of the monitoring period. Total lanthanum levels at the 4 m depth were below 0.05 mg/L post-application (Groves 2009).

The risk assessment report prepared as part of the SEPA controlled activity licence application in Loch Flemington, a host to three European protected species, included studies that indicated the frequent and regular summer blooms of cyanobacteria in the loch. From the evaluation of the external and internal sources of nutrient loading, it was identified that high levels of particulate phosphorus and chlorophyll-a were consistent with low levels of bioavailable or dissolved phosphorus. From the concentrations of the total and dissolved phosphorus in the loch, the Phoslock™ dose rate was determined as well as the optimal schedule for application (Spears & May, 2009). The schedule stated possible treatment could be done during winter to minimise effects to aquatic organisms. In March 2010, 25 tonnes of Phoslock™ were applied in the loch in a pontoon-based system. The site characteristics showed a decrease in total phosphorus levels from 74 to 37 μg/L before and after application, respectively. The levels of lanthanum were not reported (CEH, 2010).

South Africa

The Hartbeespoort Dam, frequented by dense numbers of cyanobacteria, was the site of the Phoslock™ treatment in South Africa. The study authors utilised a dosing of 250 g/m^2 to counter the effects of high pH of the dam and to provide for 1 mm sediment capping (Ross & Cloete, 2006). From the area of the water body (25000 m^2) and the dosing, the amount of Phoslock™ would be approximately 6.25 tonnes. Considering the pre-application FRP level of the dam (0.2 mg/L), this equates to a Phoslock™ to FRP ratio of approximately 417:1.

On January 2006, 6 tonnes of Phoslock™ were applied to the dam by mixing 125 kg of the granules to 1000 L of dam water in a large tank placed on a barge. The resulting slurry was then sprayed on the surface water using a hose and pump. Water samples (Phoslock™-treated and control) were collected and analysed daily (up to six days after application) and weekly (up to five weeks), and then fortnightly (up to one year). The concentrations of FRP in the treated samples decreased from 0.09 to 0.017 mg/L, on the day and after 6 days of treatment, respectively. The FRP values of the controls remained the same 6 days post-treatment. Increases in FRP levels (0.29 and 0.22 mg/L for the treated and control samples) were observed after a high rainfall event 2 weeks post-treatment. The FRP levels stabilised by week 6 with values not exceeding 0.02 mg/L in the treated site. The lanthanum concentrations were not measured in this application (Ross & Cloete, 2006).

6.3 Measures and estimates of environmental exposure

6.3.1 Methodology for measuring and estimating exposure

In accordance with the product specifications recommended by Phoslock Water Solutions (PWS), the Phoslock™ dosage rates applied directly to the water bodies are principally determined by the concentrations of FRP and the physical dimensions of the water body (total volume or surface area) (PWS, 2008d). Specifically, for large water bodies, a dosing ratio of 100:1 (on a weight basis) Phoslock™ to FRP is recommended. In principle, the initial exposure of biota in the water compartment could be estimated relatively precisely based on predetermined chemical and physical properties of the receiving water body. However, in practice, site-specific factors and operational requirements appear to dictate application rates and application methodologies for Phoslock™. The frequency of treatment also appears to be highly variable and dependent on a variety of site-specific factors including the chemistry of the receiving water body and the load of phosphate entering the water body over time (PWS, 2008d).

The estimated exposure to the environment from Phoslock™ applications to water bodies are based on the level of soluble or dissolved lanthanum, which does not necessarily correspond to the ionic or bioavailable La^{3+} found in solution.

6.3.2 Exposure data from monitoring studies

The levels of lanthanum in aquatic ecosystems and potable water supplies do not appear to be routinely monitored in Australia. A single rare earth monitoring study at a gold mining site in the Northern Territory was referred to in the *Australian and New Zealand Guidelines for Fresh and Marine Water Quality* in which the level of lanthanum in mine sump waters was found to be 3500 µg/L. The level of this element in tailings waters from this mine site was reported as 230 µg/L (ANZECC/ARMCANZ, 2000a).

6.3.3 Summary of environmental fate of Phoslock™ applications

Higher than expected concentrations of dissolved lanthanum occur in a wide variety of environmental waters 0-3 days after application of Phoslock™ granules as shown in Table 6.7. The mechanisms underlying this are a function of the various water chemistry parameters as well as the equilibrium of ionic lanthanum binding with the bioavailable phosphorus.

These data indicate that a significant proportion of the lanthanum contained in the Phoslock™ granules can be released into water under the conditions of the test (using low alkalinity (<20 mg/L) and low Filterable Reactive Phosphorus (<0.005 mg/L) water). Calculations using a range of assumptions show that the proportion of lanthanum release can be between 35% and 60%. All of the free lanthanum concentration cited in this report are measured values, and therefore the actual proportion of release lanthanum within this range does not affect the conclusions of this report.

Table 6.7. Water body characteristics and application details at the various Phoslock™ field applications.

	Initial FRP (mg/L)	Volume of water body (m^3)	Amount of Phoslock™ applied (tonne)	Calculated amount if 100:1 Phoslock™ to FRP ratio is followed (tonne)	Peak total lanthanum levels at 1-3 days post-application (µg/L)	Steady total lanthanum levels (µg/L)
Deep Creek Reservoir	0.02-0.04	4900000	55	19.6	220 (dissolved La)	n.s.
Torrens Lake	0.005	n.s.	50		110 (dissolved La)	<20 after 1 week
	<0.01		12		12 (dissolved La)	<2 after 6 days
Gnowangerup Dam No. 2	0.03	n.s.	0.3		200-250	<20 after 1 week
Barensee	0.096 & 100 kg in sediment	156000	11.5	11.5	130	<10 at 5 months monitoring
Lake Okareka	130 kg/year in sediment	n.s.	60	39	110	<10 after 1 week
Het Groene Eiland	110 kg	130000	11	11	n.s.	<30 at 5 months monitoring
Zwemplas De Kuil	n.s.	278000	41.5		56	≤20 at 10 months monitoring
De Rauwbraken	0.034-0.091 (total P)	n.s.	20		28 (dissolved La)	n.s.
Loch Flemington	239 kg	122000	25	23.9	n.s.	n.s.
Clatto Reservoir	0.079 and 0.6 in sediment	350000	24	23.8	210	<5 after 13 days

FRP = filterable reactive phosphorus; La = lanthanum; P = phosphorus; n.s. = not specified

From Table 6.7 and where data are available, most of the overseas field applications of Phoslock™ conformed to the recommended dosing ratio of 100:1 Phoslock™ to FRP. As for the Lake Okareka dosing, McIntosh (2007) stated that the annual phosphorus load on the lake also comes from external sources and septic tanks which are not included in the estimated annual phosphorus load in the sediment of130 kg/year. The lake has been monitored extensively in the three Phoslock™ applications and there were no reports of adverse effects of the applications.

6.3.4 Predicted environmental concentration (PEC)

In the original environmental risk assessment, a PEC for lanthanum in environmental waters treated with Phoslock™ was calculated as 20 µg/L. This value was calculated based on the assumption that only 0.02% of total lanthanum would be released from Phoslock™ into treated water bodies (NICNAS, 2001). However, from the actual levels of total or dissolved lanthanum from Phoslock™ field applications in environmental waters shown in Table 6.7, the peak concentrations of dissolved lanthanum significantly exceeded the original estimate of 20 µg/L, with the precise figure highly dependent on site-specific factors such as the variable rates of application of Phoslock™ to water bodies, the kinetics of lanthanum release from the chemical, and the subsequent changes in speciation of ionic lanthanum. These factors are strongly influenced by water chemistry properties including pH, alkalinity, hardness, salinity, FRP, and DOC.

The calculated amount of Phoslock™ that should have been applied to Deep Creek Reservoir based on the FRP level and recommended dosing of 100:1 is 19.6 tonnes (Table 6.7). However, the actual amount applied was 55 tonnes. Considering the low water hardness of the reservoir (27.6 mg/L $CaCO_3$) and the use of approximately three times the recommended dose, this could explain the high level of peak dissolved lanthanum of 220 µg/L.

With the currently available environmental chemistry data for Phoslock™ it is not possible to calculate a generally applicable PEC for dissolved lanthanum released into environmental waters. Consequently, the maximum concentration of dissolved lanthanum that was measured in the waters of Deep Creek Reservoir following treatment with Phoslock™ (220 µg/L) will be used as a conservative indicative figure of the peak environmental concentration of dissolved lanthanum that can occur in environmental waters after treatment with this chemical. Although the results of several laboratory tests (refer to Section 8.4) suggest that lower or higher concentrations of dissolved lanthanum may occur in waters with different chemical composition or following treatment at different application rates, the more appropriate PEC determination is from actual field application results of the product. For the purposes of risk characterisation, the peak dissolved lanthanum concentration of 220 µg/L is reasonably representative of the maximum dissolved lanthanum concentration 0-3 days following a Phoslock™ field application and will be taken forward as the PEC for lanthanum exposure in this assessment.

7. Human Health Hazard Assessment

There are no toxicological data for the notified chemical.

This section contains a short summary of the data relevant to the human health hazard assessment of the constituent chemicals, lanthanum chloride and bentonite, and the lanthanum ion La^{3+}. The toxicological data for La^{3+} are from lanthanum chloride, lanthanum nitrate, and the poorly soluble lanthanum carbonate salt. The data for these chemicals are used as surrogate data to evaluate potential human health toxicity of Phoslock™.

The data that were available and evaluated for the assessment of Phoslock™ as a new chemical are reproduced from the new chemical assessment report (NA/899) in the Appendix of this report without any modification.

Lanthanum, the main active ingredient of Phoslock™, is released in its dissolved or ionic form, which is reactive due to its high charge. The resulting lanthanum compound, dictated by the solution chemistry, will make the lanthanum available or unavailable for uptake by organs. Bioavailable lanthanum in solution is in the form of excess lanthanum ions.

Based on the reaction mechanisms of lanthanum and differences in the equilibria of the resulting compounds (Section 2.3.1), the release of ionic lanthanum in solution is more likely to occur from highly soluble than poorly soluble compounds. However, oral intake of lanthanum carbonate will solubilise this compound in the acidic stomach environment.

As there are studies documenting the release of lanthanum ions from Phoslock™ into the water body to which it is applied, the toxicological data available since the publication of the new chemical assessment report on lanthanum compounds were reviewed. Among these data are recent published studies on lanthanum based on the therapeutic use of lanthanum carbonate (Fosrenol®) in controlling hyperphosphataemia in patients suffering end stage chronic renal disease (ESRD). Reference is also made to assessment reports published by international drug evaluation agencies after review of proprietary, unpublished studies submitted by the drug sponsor.

For the Secondary Notification, several published studies on the pharmacokinetics, repeat dose toxicity, genotoxicity and developmental toxicity of lanthanum have been assessed and presented in the following sub-sections. There were no new relevant studies on bentonite.

7.1 Toxicokinetics and metabolism

7.1.1 Absorption of lanthanum (ND)

Animal studies

Oral route

The absorption of lanthanum (administered as the carbonate salt) was determined by Damment & Pennick (2007). A group of 36 male and female Sprague-Dawley rats received a single oral (gavage) dose of 1500 mg/kg bw lanthanum carbonate in a suspension with 0.5% carboxymethylcellulose (vehicle). A similar number of animals received vehicle only. Lanthanum in plasma was determined by inductively coupled plasma-mass spectrometry (ICP-MS) over 72 hours with the mean maximum concentration of 1.04 ng/mL observed at 8 hours and a decreasing 0.11 ng/mL at 72 hours.

In the same study, a group of 18 male and female Sprague-Dawley rats was administered a single dose (0.03 mg/kg) of lanthanum chloride by intravenous (iv) injection. Another group of 21 male Sprague-Dawley rats received a higher dose of lanthanum chloride (0.3 mg/kg) by iv administration. The study did not indicate any control groups for both doses. The

plasma lanthanum of the male rats given the 0.3 mg/kg dose peaked at 3231 ng/mL and rapidly decreased to 3.08 ng/mL after 48 hours. The pharmacokinetic parameters (data not presented) from the single iv administration of 0.03 mg/kg lanthanum chloride gave an area under the curve (AUC) value of 45.07 ng·h/mL from 0 to 24 hours. Oral bioavailability of lanthanum carbonate was calculated from the administration of the lower dose of lanthanum chloride and was reported to be 0.0007% (Damment & Pennick, 2007).

The drug evaluation agencies (Swedish MPA, 2006; Health Canada, 2005) found that the systemic absorption in mice and rats following oral administration of lanthanum carbonate was very low, although the values were not presented.

In a briefly reported study (Ji and Cui, 1988), rats were administered a single oral dose of a rare earth nitrate mixture (containing lanthanum nitrate) at 1000 mg/kg. At 24 hours after exposure, the absorbed fraction was measured by neutron activation analysis to be 0.066%. However, the absence of experimental detail limits the significance that can be attached to this value.

Dermal route

In an in vitro study with Wistar rat skin preparations, lanthanum nitrate was used as a tracer to evaluate the extent of oleic acid-induced skin permeation enhancement. Electron microscopy showed that the stratum corneum of intact rat skin (treated with the vehicle propylene glycol) is impermeable to water-soluble lanthanum nitrate (Jiang and Zhou, 2003).

Human studies

Oral and intravenous route

In a recent clinical trial, the bioavailability of lanthanum in healthy male volunteers was determined after administration of lanthanum carbonate and lanthanum chloride (Pennick et al., 2006). Groups of 8 volunteers received either lanthanum chloride (212 μg in 100 mL normal saline) by iv infusion over 4 hours, lanthanum carbonate (1908 mg) in a single oral tablet dose, or no treatment (control). Plasma was collected over 168 hours after start of treatment or nominal reference time (control).

Lanthanum in plasma reached a mean maximum concentration of 5.1 ng/mL at 3.3 hours after the start of the iv infusion with an elimination half-life of 37 hours. Upon oral administration, lanthanum in plasma reached a mean maximum concentration of 0.32 ng/mL at 4.5 hours after dosing with an elimination half-life of 35 hours and a concentration below the limit of detection at 48 hours. Oral bioavailability based on AUC for both oral and iv administration was calculated to be 0.00127 ± 0.00080%.

In humans, lanthanum was poorly absorbed (bioavailability <0.002%) after oral administration of lanthanum carbonate (US FDA, 2008).

Summary

The pharmacokinetic data available for lanthanum carbonate administered orally to mice, rats, and humans demonstrate very low lanthanum absorption of less than 0.002%. The oral bioavailability of lanthanum, when administered as soluble salts is likely to be higher as indicated by a single study with rats given an oral dose of rare earth nitrate mixture containing lanthanum.

In vitro studies in rats, using a soluble form of lanthanum, have shown that lanthanum does not penetrate the stratum corneum.

7.1.2 Distribution of lanthanum (ND)

Animal studies

Lanthanum carbonate

In a briefly described study designed to directly compare tissue lanthanum levels in rats receiving lanthanum via diet or gavage or by iv injection, 10 male rats per group were fed by dietary and gavage administration (0 or 838-863 mg/kg as lanthanum carbonate) or were dosed iv (0 or 30 μg/kg as lanthanum chloride) daily for 4 weeks (Damment et al., 2006). The timing and analytical method of lanthanum measurement in tissue was not reported however the authors claimed that stringent precautions were taken to ensure that the risk of contamination was minimised.

In the dietary, gavage and iv groups, plasma lanthanum levels were 0.084, 0.319 and 20.8 ng/mL, respectively, while the corresponding values in the vehicle control groups were 0.055, 0.050 and 0.069 ng/ml (lower limit of quantification = 0.05).

In all vehicle control rats, and in rats receiving lanthanum via gavage or iv injection, brain lanthanum levels were below the lower limits of quantitation (2.8 ng/g for gavage and 8 ng/g for iv injection). In rats receiving lanthanum carbonate in their diet, however, lanthanum concentrations in the cerebrum, mid-brain and cerebellum were 98.7 ± 173, 16.6 ± 20.5 and 69.8 ± 102 ng/g wet weight, despite this group having the lowest plasma lanthanum concentrations. In contrast, concentrations in liver and bone were reported as proportional to lanthanum plasma levels. As the skin concentrations of lanthanum in the diet group were dramatically higher than in those dosed by gavage or intravenously, the authors concluded that the lanthanum found in brain after dietary administration may have resulted from contamination by surface lanthanum during its removal at autopsy.

In a study conducted primarily to investigate repeat dose bone toxicity in normal renal function (NRF) and chronic renal failure (CRF) Wistar rats (Behets et al., 2004), CRF was induced in rats by nephrectomy while NRF rats were subjected to sham surgery. After 2 weeks stabilisation, NRF rats (8 to 10 males per dose) and CRF rats (7 to 10 males per dose) received 0, 100, 500 or 1000 mg/kg bw/day lanthanum carbonate by oral gavage for 12 weeks. Control groups were given the vehicle carboxymethylcellulose only. Lanthanum was measured by ICP-MS in bone at sacrifice (week 12) and plasma (4 weeks pre-treatment as baseline, 0, 4, 8 and 12 weeks). At sacrifice, femur lanthanum content for NRF and CRF animals at these dose levels was estimated to be 0.1, 0.25, 0.5, 0.6 and 0.1, 0.25, 1.25, 1.4 μg/g wet weight, respectively, with a statistically significant difference between the control and test animals in the NRF group at 500 mg/kg bw/day and above and in the CRF animals at 1000 mg/kg bw/day.

Comparing CRF versus NRF animals, a significant difference in bone lanthanum content was seen only at the top dose (data not presented). It was reported that plasma lanthanum levels showed no significant differences between the treatment groups and were independent of renal status; wherein the mean plasma lanthanum levels of NRF and CRF animals at baseline, 4 and 12 weeks were 0.44, 2.43 and 1.50 ng/mL, respectively. Similar results were obtained in a study of 20 male, Sprague-Dawley CRF rats administered 1000 mg/kg bw/day lanthanum carbonate by gavage for 6 weeks (Damment & Shen, 2005) where mean plasma and bone lanthanum levels were 1.75 ng/ml and 0.89 μg/g wet weight, respectively.

A dietary study was conducted in male Sprague-Dawley rats primarily to examine the effects of lanthanum on hepatic toxicity (Ben-Dov et al., 2007). Uraemic rats were administered lanthanum carbonate at 1.5% elemental lanthanum (10 animals) or 3.0% elemental lanthanum (15 animals) in the diet for 4 weeks. Normal and two types of uraemic rats (standard and low phosphorus diet) served as controls. Lanthanum content of plasma and liver was measured at sacrifice. Liver lanthanum levels in uraemic rats treated with 0, 1.5 and 3.0% lanthanum carbonate were 0.08, 636 and 662 μg/kg wet weight,

respectively while plasma levels were 0.05, 0.87 and 0.94 μg/L, respectively. Lanthanum was not determined in the normal and uraemic (low phosphorus diet) controls. The study indicated that in lanthanum-fed animals with compromised renal function, oral administration of high doses of poorly soluble lanthanum carbonate resulted in significantly increased levels of lanthanum in the liver. However, the increase of liver lanthanum levels was not associated with any adverse effects on liver weight or plasma levels of the liver enzymes.

Lacour et al. (2005) measured tissue distribution of lanthanum following a 28-day administration of lanthanum carbonate (0 or 3% in diet) to NRF and CRF male Sprague-Dawley rats (8 to 11 rats/group). CRF was induced by adenine overload or surgical resection. At sacrifice, lanthanum levels were higher in brain, kidney and especially liver tissue of lanthanum-fed (La[+]) NRF rats compared with NRF controls (La[-]). In rats with induced CRF there was an even greater degree of soft tissue (liver, heart, brain, lung, kidney, muscle) uptake as well as bone accumulation of lanthanum although this varied depending on the CRF model. The authors speculated that a 3- to 30-fold greater lanthanum accumulation in the liver than in other organs (greater than 1200 ng/g dry weight for both CRF groups compared with less than 400 ng/g dry weight as the highest measured in 6 other organs) was a first pass effect. There was no explanation offered for the general variations of lanthanum concentrations seen in different tissues between NRF and CRF rats, and between the two different CRF models used. Others (McLeod et al., 2005; D'Haese et al., 2005; Rambeck, 2005) have raised the possibility that the NRF to CRF and model-to-model differences are contamination-related artefacts and commented on the unusual finding of lung deposition that may reflect inhalation of aerosolized lanthanum from the high concentration, powdered diet. Dietary administration of test substances, rather than by gavage or capsule, is known to carry a higher risk of contamination. Contamination, either during the study or sample analysis, may also explain the reported lanthanum plasma concentrations that were higher in NRF and CRF control animals than the corresponding groups fed lanthanum in the diet. Overall, the possibility that contamination may have influenced the data limits the significance that can be attached to the findings of widespread tissue deposition in this rat study.

In an experiment by Slatopolsky et al. (2005), the tissue distribution of lanthanum in NRF and CRF female Sprague-Dawley rats was determined after dietary administration over 110 days. NRF and surgically-induced CRF rats receiving either 0 or 3% lanthanum carbonate (1.5% elemental lanthanum) were sacrificed at day 45 (9 to 12 rats/group), day 90 (6 to 9 rats/group) or day 110 (4 to 12 rats/group). There was a progressive rise in lanthanum content in bone, kidney and especially the liver in NRF and CRF rats fed the lanthanum diet. Uraemia enhanced tissue accumulation of lanthanum-fed rats; for the liver at day 110, means were 848, 2676, 19 and 27 ng/g wet weight for the NRF [La+], CRF[La+], NRF [La-] and CRF[La-] animals, respectively.

Lanthanum deposition in the liver did not achieve a steady state but accumulated at a steady rate over 110 days suggesting that elimination was inefficient. Compared with NRF [La+] rats, tissue accumulation of lanthanum in CRF[La+] rats was 1.5 times greater in kidney and bone. Additionally, plasma lanthanum levels were low to undetectable (data not shown) demonstrating that plasma levels are not indicative of lanthanum levels in the other tissues.

Lanthanum chloride

In a repeat dose study, male Wistar rats (6 males per dose) received 0, 0.1, 2 or 40 mg/kg bw/day lanthanum chloride by gavage for 6 months (Feng et al., 2006b). Content of lanthanum in the brain was measured by ICP-MS. Quantifiable lanthanum was absent in 4 regions of the brain at 0.1 and 2 mg/kg bw/day however, at 40 mg/kg bw/day, lanthanum content in the cerebellum (15 ng/g dry weight), cerebral cortex (39 ng/g) and the rest of the brain (20 ng/g) was statistically higher than in the controls (<5 ng/g).

In a neurodevelopmental study (He et al., 2008), Wistar rats (10 mated females/group) were administered lanthanum chloride by gavage at 0, 0.1, 2 and 40 mg/kg bw/day on gestation day 0 to postnatal day 20 (i.e. the end of weaning). At weaning, male pups (40/group) were separated from the dams and gavaged at the same doses until sacrifice at six months. Lanthanum concentrations in the serum, hippocampus and cerebral cortex were measured by ICP-MS (6/group). In rats fed 40 mg/kg bw/day, lanthanum was detected in the hippocampus (14 ng/g dry weight) and in the cerebral cortex (40 ng/g) at statistically higher levels than in the controls (<4 ng/g for both tissues).

The appearance of lanthanum in liver cells as a result of intravenous injection of lanthanum chloride was investigated by advanced transmission electron microscopy (Yang et al., 2006). Rats (sex and number not reported) were administered 0.3 mg/kg bw/day lanthanum chloride for 4 weeks. Lanthanum seen in liver sections was present as a granular precipitate (ranging in concentration and size from 30 to 50 μg/g and 5 to 25 nm, respectively) confined solely to lysosomes that were preferentially localized near the bile canaliculi.

Lanthanum nitrate

In a briefly reported study (Ji and Cui, 1988), lanthanum absorbed by rats administered a single oral dose of a rare earth nitrate mixture (containing lanthanum nitrate) was found predominantly in the bones and liver followed by the heart and kidneys.

Drug agency evaluations of lanthanum carbonate

Lanthanum does not accumulate in plasma at repeated oral dosing of lanthanum carbonate. In vitro, lanthanum is extensively (>99%) bound to mouse, rat, rabbit and dog plasma proteins. In other studies in these species, lanthanum accumulation in many tissues is seen but particularly in the gastrointestinal tract, stomach, bone and liver. The chemical form(s) of lanthanum in the tissues is unknown. An apparent steady state was achieved in some tissues (bone and liver) in dogs, however relatively high levels were still measurable six months after cessation of treatment.

The liver and spleen were the primary sites for deposition of lanthanum after repeated iv administration of soluble lanthanum chloride (Health Canada, 2005; US FDA, 2008; Swedish MPA, 2006).

Human studies

Lanthanum carbonate

A study in dialysis patients monitored plasma and bone lanthanum levels during 1-year oral administration of lanthanum carbonate and following a 2-year washout period (Spasovski et al., 2006). Six male and 4 female patients (aged from 45 to 65 years old) received a titrated oral dose of up to 3000 mg lanthanum carbonate per day over 1 year and then maintained on a standard calcium carbonate therapy for 2 years. A second group of 6 male and 4 female patients (aged from 47 to 67 years old) received a titrated oral dose of up to 4000 mg calcium carbonate per day over 3 years.

During lanthanum carbonate treatment, plasma lanthanum levels increased to a mean maximum of 1.26 ng/mL at 24 wks, stabilized at approximately 0.6 ng/mL from 36 to 52 weeks (cessation) and significantly declined to 0.17 ng/mL at 6 weeks of washout. There was no further significant decrease during the remainder of the 2-year washout with final levels of 0.09 ng/mL which was significantly higher than those patients in the calcium carbonate control group (<0.05 ng/mL). Plasma lanthanum levels did not correlate with the dose at any time point.

The concentration of bone lanthanum increased in all patients during the trial with mean bone concentrations at 1 year for the lanthanum and calcium groups significantly different at 2.3 and 0.1 μg/g, respectively. At the end of the washout period, the lanthanum group had decreased slightly, but significantly, to 1.9 μg/g with the calcium group increasing to 0.15 μg/g.

While there was no correlation between the bone lanthanum content and dose at any time point, there were significant correlations between bone lanthanum (at the end of the 1-year study) and plasma lanthanum at the 1- and 2-year washout and bone lanthanum (at 2-year washout) with plasma lanthanum levels at 1 year.

In a similar study, 34 dialysis patients were treated with lanthanum carbonate (up to 3750 mg/day) and 34 patients with calcium carbonate (up to 9000 mg/day) for 1-year (D'Haese et al., 2003). At 1 year, mean plasma lanthanum levels in the lanthanum group were slightly increased, although independent of dose, ranging from 0.51 to 1.08 ng/mL having plateaued after 12 weeks of treatment and mean bone levels were 1.8 µg/g.

Summary

Animal studies have shown that absorbed lanthanum is widely distributed to tissues many of which retain lanthanum at levels significantly above that of plasma. Accumulation during treatment is reported, particularly in bone, liver, stomach and other gastrointestinal tract tissues with relatively high levels of lanthanum (chemical form(s) unknown) remaining in these tissues for an extended period after cessation of dosing.

There are insufficient data regarding the ability of lanthanum to cross the blood-brain barrier. While limited rat studies using lanthanum carbonate have reported detection in brain tissue following oral administration, it appears that contamination by extraneous lanthanum may be responsible for the discrepant tissue deposition profiles. Lanthanum in the brain was not detected in rats intravenously administered lanthanum chloride.

In clinical studies, accumulation of lanthanum in the bone of dialysis patients was evident during oral administration of lanthanum carbonate and the dissipation following cessation of the treatment was very slow. However, no correlation was established between the lanthanum levels in the bone and the administered dose even after the washout period. Due to the observed variability of results and the specific conditions with the renal function in the subjects these studies are of limited value in assessing tissue distribution of lanthanum in the general population.

7.1.3 Elimination and excretion of lanthanum (ND)

Animal studies

A number of experiments on absorption and excretion of lanthanum are reported in a well conducted study by Damment & Pennick (2007). Within 24 hours of oral gavage administration of 600 mg/kg bw lanthanum carbonate to 4 male Sprague-Dawley rats, a mean of 97.8% of the administered dose had been excreted over 168 hours. The primary route of excretion was through the faeces (mean of 99.4% of the dose) with urine being a minor route (0.0035%).

A group of 12 male rats from the above study received either 0.3 mg/kg bw lanthanum chloride or saline by a single iv injection. The very low solubility of lanthanum carbonate precludes its suitability for iv administration. Over 42 days, 76.4% of the administered dose had been recovered with the major route of excretion through the faeces (96.9%) and urine being a minor route (1.94%). Most faecal lanthanum (87.7%) was excreted during the first 14 days with 41% of the urinary lanthanum excreted during the first 24 hours.

Similarly, in dogs, the mean recovery of lanthanum after an oral dose of lanthanum carbonate was 94% and was essentially all recovered from the faeces (US FDA, 2008).

Human studies

Lanthanum chloride

The excretory routes of lanthanum in healthy volunteers were determined after intravenous lanthanum administered as the soluble chloride salt (Pennick et al., 2006). Eight Caucasians received lanthanum chloride (212 µg in 100 mL normal saline) by iv infusion over 4 hours.

Blood, urine and faeces were collected over 168 hours after start of treatment or nominal reference time (control).

Lanthanum in plasma reached a mean maximum concentration of 5.1 ng/mL at 3.3 hours after the start of the iv infusion with an elimination half-life of 37 hours. Over 168 hours after initiation of the lanthanum chloride infusion, 1.75% of the dose was excreted in the urine, with more than 90% of this eliminated during the initial 72 hours. Faecal excretion of lanthanum could not be quantified above the high and variable background levels that were present due to the natural occurrence of lanthanum in many foods.

In humans, lanthanum was excreted via nonrenal mechanisms as indicated by the negligible renal clearance rates in the study.

Summary

Overall, in the rat, dog and human, systemic lanthanum is predominantly excreted via the faeces, regardless of the route of administration, with minimal excretion in the urine.

7.2 Effects on laboratory animals and other test systems

7.2.1 Acute toxicity

No new toxicological data were provided for Phoslock™ or the similar lanthanum salts. Based on the data for the two ingredients, lanthanum chloride and bentonite, Phoslock™ is expected to have very low acute oral toxicity in humans (NICNAS, 2001).

7.2.2 Irritation and sensitisation

Phoslock™ may cause mild transient physical irritation on eye contact due to the presence of clay particles and is not expected to be a skin irritant (NICNAS, 2001).

No new information is available on the irritation and sensitisation potential of Phoslock™ or its constituents.

7.2.3 Repeated dose toxicity (ND)

The original assessment of Phoslock™ concluded that its toxicity profile likely resembles that of bentonite. Based on the 33-day repeated oral administration of bentonite to chickens, decreased growth, muscle weakness, changes in calcium and phosphorus metabolism, and death were reported at doses higher than 6% (NICNAS, 2001).

Lanthanum carbonate

A 12-week study was conducted primarily to investigate the effects of lanthanum carbonate on bone histology in normal and uraemic rats (Behets et al., 2004).

Normal renal function (NRF) Wistar rats (8 to 10 males per dose) and chronic renal failure (CRF, surgical resection) rats (7 to 10 males per dose) received 0, 100, 500 or 1000 mg/kg bw/day lanthanum carbonate by gavage for 12 weeks. Bone histomorphometry was assessed using traditional light microscopy.

An impairment of bone mineralisation (as measured by a decrease in bone formation rate and increase in osteoid area) was seen in 3/7 (43%) of the 1000 mg/kg bw/day CRF group and 1/8 (13%) of the 100 mg/kg bw/day NRF group, with other animals in all groups presenting normal bone histology. Additionally, there was no correlation between the bone lanthanum concentrations measured and development of a mineralisation defect and no evidence that lanthanum had a direct toxic effect on osteoblast number and activity. The authors determined the LOAEL, based on increased bone lanthanum content, in normal and uraemic animals to be 1000 and 500 mg/kg bw/day, respectively. The significance that can be attached to the LOAEL of uraemic rats is limited since marked increases in osteoblast perimeter (indicating presence of active osteoblast formation) were observed at 0 and 100

mg/kg bw/day but were not observed at 500 and 1000 mg/kg bw/day doses suggesting that the LOAEL could be lower based on the histological findings not seen at the higher doses.

Further studies produced evidence that the mineralisation defects were pharmacologically mediated and the result of phosphate depletion rather than direct toxicity. Damment & Shen (2005) reported that the effects of lanthanum on bone mimicked those induced by feeding rats a low phosphate diet and that the effects of lanthanum were normalised with phosphate repletion (Behets et al., 2005).

In a 4-week study, the effect of lanthanum on hepatic toxicity in male Sprague-Dawley rats was investigated (Ben-Dov et al., 2007). Uraemic (induced by adenine overload) rats were administered lanthanum carbonate at 1.5% elemental lanthanum (10 animals) or 3.0% elemental lanthanum (15 animals) in the diet for 4 weeks. Bodyweights and clinical observations were recorded. Based on a reported mean animal weight of 0.325 kg and a default food consumption of 0.018 kg/day, daily lanthanum intakes were estimated to be 830 and 1660 mg/kg bw/day, respectively. Normal, uraemic (standard diet) and uraemic rats (low-phosphorus diet) served as controls (20, 20 and 5 animals, respectively). During week 3, liver-oriented magnetic resonance imaging (3 to 5 rats per group) revealed no abnormalities in any group. At necropsy, assessment of plasma liver enzymes (aspartate transaminase, alanine transaminase, alkaline phosphatase and gamma-glutaryl transferase), histology and assessment of liver weights revealed no evidence of adverse effects on the liver. Under the conditions of the study, the NOAEL for liver toxicity was considered to be 1660 mg/kg bw/day.

Repeat dose oral studies reported in a conference presentation abstract (Webster & Jones, 2004) investigated the effects of lanthanum carbonate up to 2,000 mg/kg bw/day in mice (up to 99 weeks duration), rats (up to 154 weeks) and dogs (up to 52 weeks), together with shorter term iv studies using the soluble chloride salt. The study claimed that there was no functional or histopathological evidence of long term central nervous system toxicity. The validity of these findings is difficult to assess as the methodological details were not described fully.

Lanthanum chloride and lanthanum nitrate

A study translated from Japanese (Ogawa et al., 1992) was available as a summary from a secondary source (TERA, 1999).

Lanthanum was tested in Wistar rats (5 animals/dose/sex) at doses of 0, 40, 200, or 1000 mg/kg bw/day (as $LaCl_3 \cdot 7H_2O$) administered by gavage for 28 days. During lanthanum exposure, the body weight decreased (10-15% compared to control) in both sexes at 200 mg/kg bw/day and above, but they were comparable to that in the pair feeding group, suggesting the change in bodyweight gain is related to food consumption. Haematological examination indicated an increase in eosinophils in females at 40 mg/kg bw/day and above, and in males at 200 mg/kg bw/day and above. However the significance of this change is unknown. At 200 mg/kg bw/day and above, a significant decrease of serum cholinesterase activity was observed only in females while serum transaminase activity increased in both sexes at 1,000 mg/kg bw/day. The latter finding was suggestive of hepatotoxicity, although no corresponding histopathological changes in the liver were reported. The histopathological data indicated significant increases in lung granulation and giant cell appearance at 200 mg/kg bw/day and above. As discussed in a later paper (Ogawa et al., 1995 cited in TERA, 1999), the authors suggested that these changes in the lungs might be caused by the inhalation of the test compound. In addition, 1,000 mg/kg bw/day resulted in stomach lesions, including infiltration in the submucosa in sexes, erosion and dilatation of the acinus in males and swelling in the epithelium in females.

Based on changes in liver enzyme activity levels at 200 mg/kg bw/day, the NOAEL was judged to be 40 mg/kg bw/day.

A six-month oral study was conducted to investigate brain biochemistry and neurotoxic effects of lanthanum in male rats (Feng et al., 2006b). Groups of 15 Wistar rats were gavaged with lanthanum chloride at a dose equivalent to 0, 0.1, 2, and 40 mg/kg bw/day from age 4 weeks for 6 months. At the end of the treatment period, 12 animals per group were trained and assessed over 8 days in a Morris water maze test and then sacrificed. Compared with controls (latency times 25 seconds at day 2, 22 seconds at day 3, and 21 seconds at day 4), a statistically significant difference in escape latency times was seen in the 40 mg/kg bw/day group over days 2 to 4 (latency times 35 seconds at day 2, 32 seconds at day 3, 30 seconds at day 4) after which all groups were equivalent from day 4 to day 8.

Synchrotron radiation X-ray fluorescence (SR-XRF) images used to examine the distribution of calcium, iron and zinc in brain sections showed marked decreases in metal (Ca, Fe, and Zn) concentrations in specific brain regions at 2 mg/kg bw/day and above. Additionally, compared with controls, almost complete depletion of calcium and zinc at mid- and high-dose levels was evident in the hippocampus, cerebral cortex, and thalamus. At the top dose of 40 mg/kg bw/day, lanthanum content in the cerebellum (15 ng/g dry weight), cerebral cortex (39 ng/g dry weight) and the rest of the brain (20 ng/g dry weight) was statistically higher than the controls (up to <5 ng/g dry weight in the cerebellum, cerebral cortex, and in the rest of the brain).

A statistically significant decrease (12%) in brain Ca^{2+}-ATPase activity was seen at 40 mg/kg bw/day compared with untreated controls, while approximately 12% decrease in acetylcholinesterase activity and 11% increase in acetylcholine concentration were observed at the lowest dose only.

The levels of monoamine neurotransmitters were evaluated by HPLC with Electrochemical Detection (HPLC-ECD). Statistically significant, dose-dependent reductions in the level of brain neurotransmitters dopamine, dihydroxyphenylacetic acid and noradrenaline were observed at 40 mg/kg bw/day (22, 22 and 28%, respectively). Marked decreases in 5-hydroxytryptamine at 2 mg/kg bw/day and 40 mg/kg bw/day (up to 23% for each dose group) were observed, however, dose dependence was unclear. Reductions in the concentrations of homovanillic acid and hydroxyindoleacetic acid of 11 and 8%, respectively, failed to achieve significance and clear dose dependence at all dose groups could not be established.

The effects in the escape latency times at 40 mg/kg bw/day may be attributed to the decreases in the neurotransmitter levels. However, significant differences in the escape latency times at the highest dose were observed in days 2 to 4 but recovered from days 4 to 8. Changes in the levels of neurotransmitters are not necessarily indicative of neurotoxic effects. However, if well correlated with differences in neurobehaviour, the changes could be justified as contributing to neurotoxicity (US EPA, 1998). In this study, a clear association between the neurotransmitter levels and the escape latency times was not shown.

Therefore, the NOAEL was considered to be 2 mg/kg bw/day based on effects on brain biochemistry (a decrease in dopamine and noradrenaline levels and alterations to brain elemental distribution) observed at 40 mg/kg bw/day.

A dietary study was conducted to investigate the biochemical and histological effects of lanthanum on the liver (Chen et al., 2003). Wistar rats (15/sex/group) were gavaged with lanthanum nitrate with a dose equivalent to 0, 0.1, 0.2, 2.0, 10.0 and 20.0 mg/kg bw/day at 6 times per week for 6 months.

Body weight gain in male rats at the top dose was significantly reduced compared with controls in contrast to a significant increase at the lowest dose (though the bodyweight change levels were not reported). Differences in relative liver weights did not show any consistency or dose-relationship to indicate that they were related to exposure. There were histological changes in the liver at the top dose with infiltration of inflammatory cells in the

portal area, reduction in glycogen granules, irregular nuclear outline and lipid droplets in hepatocytes compared with controls. Additionally at the top dose, and present at reduced number at lower dose levels, lysosomes containing electron dense particles were visualised by transmission electron microscopy in hepatocytes distributed predominantly around bile canaliculi.

Differences in the activities of the enzymes superoxide dismutase, glutathione peroxidase, alkaline phosphatase and the oxidative stress marker, malondialdehyde, were not dose-related while the serum levels of glutamic-oxalacetic transaminase, glutamic-pyruvic transaminase and gamma-glutamyl transferase were no different to controls. Overall, a NOAEL of 10 mg/kg bw/day was determined based on a reduction in body weight gain and histopathological changes in the liver seen at the top dose. However, there is a significant level of uncertainty associated with this NOAEL due to poor reporting of this study.

Drug agency evaluations of lanthanum carbonate

Repeat dose oral studies of lanthanum carbonate at 2000 mg/kg bw/day in dogs (52 weeks dosing) established accumulation in the liver, bone and gastrointestinal (GI) tract. Chronic hepatitis developed in dogs given 1 mg/kg bw/day lanthanum chloride intravenously for four weeks. In long-term toxicity studies in rats, the increased likelihood of osteomalacia (bone demineralisation or softening) was dependent on increasing lanthanum carbonate levels (Health Canada, 2005).

Time and dose-dependent increases in histopathological changes in the stomach were seen in rodents but not in dogs. The Swedish Medical Products Agency (MPA) (2006) established a NOAEL of 100 mg/kg bw/day in the mouse for epithelial hyperplasia, hyperkeratosis of the limiting ridge and mucosal cell infiltration. The same NOAEL was established in the rat for these effects together with mineralisation, mucosal inflammation and eosininophilia of chief cells. In the absence of gastric changes in the dog, the stomach toxicity may be related to the unique physiology of the rodent stomach and therefore the clinical significance of these changes may be limited.

Hepatotoxicity, as evidenced by cellular infiltration and chronic hepatitis in rats and dogs, respectively, was seen predominately in iv studies (lanthanum chloride) when high tissue concentrations were reached and little evidence of any liver effects was seen in oral studies using lanthanum carbonate (Swedish MPA, 2006).

Repeated oral administration of lanthanum carbonate (up to 2000 mg/kg bw/day) to dogs (up to 52 weeks) (Health Canada, 2005; Swedish MPA, 2006), rats (up to 26 weeks), and mice (up to 13 weeks) (Swedish MPA, 2006) showed no evidence of treatment-related bone toxicity.

Summary

Repeat dose oral toxicity studies for lanthanum carbonate in the rat, mouse and dog, and supplemental iv studies on lanthanum chloride, have established the stomach and liver as primary targets for lanthanum toxicity.

Stomach inflammation and hyperplasia found in rodents at a NOAEL of 100 mg/kg bw/day, were considered to be rodent specific since the same effects were not observed in dogs. Elevated levels of lanthanum in dogs were not associated with visible toxicity effects at 2000 mg/kg bw/day. Also, hepatotoxicity was evident in animals administered soluble lanthanum by the iv route but was not consistently observed in oral studies with lanthanum carbonate.

There was no evidence that lanthanum carbonate induced any direct bone toxicity despite the bone being a principal site of lanthanum deposition.

In the rat, two oral toxicity studies for the soluble lanthanum chloride and nitrate salts have commonly demonstrated liver effects and a trend for reduced body weight gain. Neurotoxicity, as demonstrated by changes in brain neurotransmitter levels, trace element

distribution in the brain and impaired cognitive ability, was seen in a 6-month rat study using lanthanum chloride. A LOAEL for these brain alterations and learning decrements was reported to be 40 mg/kg bw/day, and the NOAEL was 2 mg/kg bw/day.

7.2.4 Genotoxicity (ND)

Lanthanum chloride was non-mutagenic in a bacterial mutagenicity study while a reliable conclusion could not be drawn about the mutagenic and clastogenic potential of lanthanum chloride administered by intraperitoneal injection in rat and mouse bone marrow cells (NICNAS, 2001).

Bacterial reverse mutation test using lanthanum carbonate

In a bacterial (*Salmonella* and *E.coli*) point mutation assay (Damment et al., 2005), lanthanum carbonate doses ranging from 8 to 5000 μg/plate showed no increase in the incidence of gene mutations in *Salmonella* strains TA 98, TA 100, TA 1535 and TA 1537, TA 1538 or in E. coli strain WP2 uvrA with or without metabolic activation (S9). However, with E. coli strain WP2 uvrA (pkm101) there was a <2-fold increase in revertants observed in the first experiment at 500 and 2500 μg/plate with or without metabolic activation. The increases were not dose-related or reproducible in a second independent experiment and were within the historical control range.

There was evidence of cytotoxicity in all test strains, with or without activation, at the top dose, and controls gave results in the expected ranges. It was concluded that lanthanum carbonate was not mutagenic under the conditions of the test.

In vitro mammalian gene mutation test using lanthanum carbonate

Lanthanum carbonate was evaluated for its ability to induce *hgprt* gene mutations in Chinese hamster ovary (CHO) cells according to OECD TG 417 (Damment et al., 2005). There were no significant effects on mutation frequency at doses up to the maximum of 5000 μg/mL at 3 hours with S9 in the absence of cytotoxicity. Dose independent, non-reproducible increases in mutations were seen at 500 and 2000 μg/mL at 3 hours without S9 and in the presence of cytotoxic effects.

Controls gave results that confirmed the validity of the test and the test compound was not determined to be a mutagen.

In vitro cytogenetics assay using lanthanum carbonate

In a chromosome aberration study consistent with OECD TG 476 (Damment et al., 2005), Chinese hamster ovary (CHO) cells were treated with lanthanum carbonate at concentrations up to 550 μg/ml for 24 hours without S9 and at concentrations up to 5000 μg/ml for 3 hours with S9. Cells were harvested 24 or 48 hours after treatment initiation. A small but statistically significant increase in the incidence of cells with aberrant chromosomes was seen from 350 μg/mL and above at 24 hours without S9 (2/3 independent experiments) and at 5000 μg/mL at 24 and 48 hours with S9 (2/2 experiments). However, the authors reported that the increases in the frequency of chromosomal aberrations fell within the historical control range for the cell line and were only apparent at lanthanum carbonate concentrations that were marginally cytotoxic and which resulted in considerable precipitation (200 μg/ml and above).

Overall, the possible cytotoxicity and confounding effects of precipitation means no reliable conclusions can be drawn from the data on the clastogenic potential of lanthanum carbonate.

In vivo mammalian bone marrow micronucleus test after administration of lanthanum carbonate and lanthanum chloride

Lanthanum carbonate had no effect in a micronucleus assay in CD-1 mice (5/sex) administered single gavage doses of 800, 1250 and 2000 mg/kg bw (Damment et al., 2005).

Vehicle and positive control groups received 0.5% carboxymethyl cellulose and 4 mg/kg bw mitomycin C (ip), respectively, with bone marrow sampled at 24, 48 and 72 hours.

In a further micronucleus assay by Damment et al. (2005), lanthanum chloride was iv administered to 6 male Sprague-Dawley rats at doses of 0.025, 0.05 or 0.1 mg/kg bw which were chosen as representative of plasma exposures encountered in clinical use of orally administered lanthanum carbonate. Vehicle and positive control groups received 0.9% sodium chloride (iv) and 20 mg/kg bw cyclophosphamide (oral), respectively, with bone marrow sampled at 24 and 48 hours. All treatment and vehicle controls exhibited normal frequencies of micronucleated polychromatic erythrocytes, while the positive control gave a clear increase in the frequency of micronuclei. In conclusion, lanthanum chloride does not induce micronuclei in rat bone marrow after iv administration of up to 0.1 mg/kg bw.

In vivo unscheduled DNA synthesis (UDS) test in rat liver after iv administration of lanthanum chloride

In an unscheduled DNA synthesis (UDS) test in rat liver in vivo conducted according to OECD TG 486 (Damment et al., 2005), lanthanum chloride was administered to 3 male Han Wistar rats per dose as iv injection of 0.025, 0.05 or 0.1 mg/kg bw/day for 28 days. Use of this multiple dose schedule ensured maximum exposure of the liver to lanthanum therefore allowing a more rigorous evaluation of its genotoxic potential. Vehicle and positive control groups received 0.9% sodium chloride (iv) and 20 mg/kg bw cyclophosphamide (oral), respectively. Hepatocytes isolated 12-14 hours after the last lanthanum chloride dose did not show any indication of UDS at doses up to 0.1 mg/kg bw/day whereas the positive control produced clear increases in UDS.

Summary

Lanthanum carbonate had no effect in a bacterial reverse mutation test. In mammalian cells in vitro, conflicting results were observed. In a gene mutation and a chromosome aberration study, a negative and an equivocal result, respectively, have been reported although the increase in aberration frequency was within the historical control range and apparent at marginally cytotoxic concentrations that were at the solubility limit.

In several in vivo tests, no genotoxic effects were observed in parallel bone marrow chromosome aberration studies (using lanthanum carbonate and lanthanum chloride) and a UDS study using lanthanum chloride.

Overall, there are a number of well conducted in vitro and vivo studies available and the weight of evidence indicates that lanthanum carbonate and lanthanum chloride are not genotoxic.

7.2.5 Carcinogenicity (ND)

No published primary data are available.

Drug agency evaluations of lanthanum carbonate

Oral administration of lanthanum carbonate to mice and rats for up to 104 weeks, under GLP conditions, was conducted at doses up to 1500 mg/kg bw/day (Swedish MPA, 2006). The type of oral administration was not specified in the report but is likely to be by gavage. Treatment at 1500 mg/kg bw/day was associated with an increased incidence of glandular stomach adenomas in male mice, but not in rats.

The neoplastic stomach changes in the mouse were subjected to extensive review and evaluation by pathologists who concluded that they were likely to be related to an age-related exacerbation of spontaneous pathological stomach changes in CD-1 mice. As such, and given the negative findings in rats, the stomach adenomas in this sensitive mouse strain are probably related to local irritant effects and have little clinical significance (US FDA, 2008; Swedish MPA, 2006).

7.2.6 Reproductive toxicity (ND)

The original assessment of the notified chemical found that lanthanum chloride administered to goats caused adverse effects on spermatogenesis and intraperitoneal injections of the chemical to pregnant mice reduced the number of successful pregnancies and litter size (NICNAS, 2001).

Developmental toxicity of lanthanum chloride in mice

The effect of lanthanum on neurodevelopment was evaluated in a drinking water study in mice (Briner et al., 2000). Female Swiss-Webster mice (group sizes not reported) were administered lanthanum chloride heptahydrate in the drinking water at 0, 125, 250 and 500 mg/L (estimated at 0, 11, 23 and 45 mg/kg bw/day based on a body weight of 0.03 kg and water consumption of 6 mL/day) for 14 days prior to mating. Exposure to lanthanum continued through gestation, parturition and the postnatal period for the dams and pups (birth day designated postnatal day 0) until sacrifice at postnatal days 59-60. Assessments of behavioural and neurologic parameters were made from days 4 to 20 and 30 to 32 and brain chemistry was assayed at sacrifice. Group sizes for the assessments of pups were not reported.

No treatment related clinical signs of toxicity or differences in litter size were seen in dams. Additionally, no effects were seen on mortality or general health of pups at birth or weight gain of pups over the postnatal period day 4 to 20.

Eye opening at day 14 was significantly delayed for the 11 and 23 mg/kg bw/day groups compared with controls, however this effect was not dose-related as there was no delay observed for the top dose group. At all other ages, variance in eye opening was negligible. Similarly, there were observations (not statistically significant) of ear opening being delayed at day 13 for the 11 and 23 mg/kg bw/day groups but not at the top dose. The emergence of walking behaviour was significantly delayed at days 5 and 6 for the 23 mg/kg bw/day group only. Additionally, there was a statistically significant delay in swimming development for postnatal days 6 to 14 for the 23 and 45 mg/kg bw/day group. While changes were observed (days 30 to 32) in touch response and visual placing response performance, they were not dependent on dose.

At sacrifice, litter body weight and brain lanthanum, protein and lipid content was consistent among groups. There was a small, but statistically significant, difference in absolute brain weight at 45 mg/kg bw/day compared with controls with no difference in relative brain weight.

In this study, lanthanum chloride did not appear to exhibit maternal toxicity up to 45 mg/kg bw/day and, although scattered observations of delayed neurodevelopment of offspring have been observed, there was no dose-response relationship established for eye and ear opening, walking, touch response and visual placing response.

Consequently, overall, it is considered that the NOAEL is 11 mg/kg bw/day based on neurobehavioural effects observed from a statistically significant delay in the emergence of swimming behaviour of the offspring starting at the mid dose level of 23 mg/kg bw/day. However, the failure to demonstrate dose-response relationships for the other neurobehavioural parameters investigated in this study limits the significance that can be attached to the data.

158 day developmental toxicity study of lanthanum chloride in rats

A long-term neurodevelopmental study on the effects of lanthanum chloride was conducted in rats (Feng et al., 2006a). Wistar rats (15 mated females/group) were orally administered lanthanum chloride at 0, 0.1, 2 and 40 mg/kg bw/day on gestation day 0 to postnatal day 20 (i.e. the end of weaning). At weaning, pups (60/sex/group) were separated from the dams and exposed to lanthanum at the same doses until interim sacrifice at postnatal day 30 (10 males/group) or completion of behavioural testing using the Morris water maze test at day 158 (12 males and 6 females/group).

There were no treatment related effects on litter size, pinna detachment or eye opening. Maternal toxicity was not reported. Body weights in offspring were measured at day 0, 30, 90 and 150. A statistically significant increase in body weight gain (7%) was seen at day 90 only in males receiving 2 mg/kg bw/day and a decrease in weight gain in males and females (11 and 8%, respectively) at the top dose at day 150. Surface righting reflex was significantly faster for the mid and top dose groups at day 3 (36 and 34%, respectively) and day 4 (18 and 15%, respectively) but not different on day 5. At day 20 the swimming time for the mid dose group was facilitated (28%) however, in contrast, at the top dose it was impaired significantly compared with controls (43%). No differences in swimming time were seen among groups at day 10.

At day 150, the escape latency of the (now adult) 40 mg/kg bw/day group was significantly increased by 94-136% from day 3 to day 5 in the 8-day training regime compared with controls.

At interim sacrifice (day 30), there was a slight, but significant, decrease in the brain DNA content at the mid and top doses (data presented graphically only) with no dose related effect on DNA/protein ratio observed.

Overall, the scattered and inconsistent results seen in body weight gain and the three learning and memory tests limit the significance that can be attached to the neurobehavioral findings in this study.

Six month developmental toxicity study of lanthanum chloride in rats

In a neurodevelopmental study (He et al., 2008), Wistar rats (10 mated females/group) were administered lanthanum chloride by gavage at 0, 0.1, 2 and 40 mg/kg bw/day from gestation day 0 to postnatal day 20 (i.e. the end of weaning). At weaning, male pups (40/group) were separated from the dams and gavaged at the same doses until six months of age. Assessments of behavioural performance on the offspring were made using the Morris water maze test at 6 months of age. Brain histopathology, biochemistry and lanthanum distribution were also examined following necropsy.

This study does not report any parameters related to effect of treatment on dams.

There was no significant difference in the body weight among the animals used in the behavioural Morris water maze test. In the training test, significant increase of mean escape latency was observed in the animals at the mid and top dose group for the 3rd and 4th training sessions compared with controls. The length of mean general pathway in finding the escape platform for rats of the top dose group was also significantly prolonged relative to the controls in the 4th training test. The results indicate that lanthanum exposure at 2 and 40 mg/kg bw/day is associated with impaired acquisition of spatial learning and memory. In the trial test assessing consolidation of memory, animals from the 2 and 40 mg/kg bw/day group spent 16.6% and 19.4% less time, respectively, in the target quadrant of the maze compared with controls. Histopathology examination of the hippocampus of animals from the mid and high exposure group showed statistically significant decrease in pyramidal cells in the CA3 subregion by 18% and 23%, respectively. Other regions of the hippocampus were not affected.

Statistically significant increase of free intracellular calcium levels and decrease of Ca^{2+}-ATPase activity was observed in the hippocampal cells of rats in the high dose group compared with controls.

A statistically significant increase in malondialdehyde, an indicator of increased lipid peroxidation in the cells, was seen in the hippocampus (top dose) and the cerebral cortex (mid and top dose) compared with controls. This correlated with statistically significant decrease of the activity of catalase in hippocampal tissue at the mid and top dose, superoxide dismutase in cerebral cortex tissue at the top dose and glutathione peroxidase in both tissues of top dose rats. It is unclear whether these biochemical changes precede or follow the change in cell numbers in the hippocampal C3 region.

Lanthanum concentrations in the serum, hippocampus and cerebral cortex were measured by ICP-MS (6/group). In rats fed 40 mg/kg bw/day, lanthanum was detected in the hippocampus (14 ng/g dry weight) and in the cerebral cortex (40 ng/g dry weight) and in the serum (0.6 ng/mL) at statistically higher levels than in the controls (3 and 4 ng/g dry weight for hippocampus and cerebral cortex, respectively; and <0.1 ng/mL for serum).

In this study, neurodevelopmental toxicity was evident at 2 mg/kg bw/day, with statistically significant impairment seen in spatial learning and memory abilities that are correlated with decreases in the number of pyramidal cells in the C3 subregion of the hippocampus and biochemical changes in the hippocampus and cerebral cortex indicating oxidative stress and perturbations in the antioxidant enzyme activity. Therefore, the NOAEL for developmental toxicity was determined to be 0.1 mg/kg bw/day.

Subchronic developmental neurotoxicity study of lanthanum chloride in rats

In a 7-week neurodevelopmental study (Yang et al., 2009) lanthanum chloride was administered via distilled drinking water to Wistar rats. Lactating dams were exposed to 0%, 0.25%, 0.5% and 1.0% lanthanum chloride for three weeks. Eight litters (8-10 pups) were allocated per dose. After weaning, the pups were orally administered lanthanum chloride in distilled water at the same concentrations as the dams (0%, 0.25%, 0.5% and 1.0% $LaCl_3$ or 0, 250, 500 and 1000 mg/kg bw/day $LaCl_3$ approximately) for one month. Effects on memory were investigated using water maze test (10 trials/day for 5 days) and by inspection of the synaptic ultrastructure of hippocampal CA1 region using transmission electron microscopy. The effects of lanthanum on expression in the hippocampus of protein kinases involved in maintaining memory such as pCaMK IV, pMAPK, pPKA and pCREB was also investigated by western blot analysis.

A significant dose-related increase in lanthanum concentration in the hippocampus was observed in all dose groups with 25 times higher at the top dose compared with controls. From training days 2-4 escape latency was significantly longer in all treated groups but no difference was observed at day 5. After 1 week of rest, a significant dose-related increase of escape latency was seen in all treated groups. This indicated weaker spatial memory in dosed rats compared with controls. The integrated density values for pCaMK IV, pMAPK and pCREB in the hippocampus were significantly lower than controls in all dose groups (each dose significantly lower than the previous dose). No dose-related response was seen in pPKA expression. Photomicrographs of the ultrastructure of synapses in the CA1 region showed abundance in synaptic vesicles, a long active synaptic zone, smooth synaptic curvature and thick post-synaptic density in control rats. These were not observed in all of the treated groups.

No NOAEL for developmental toxicity could be derived from this study. The LOAEL is 250 mg/kg bw/day due to effects in memory, expression of protein kinases associated with memory and altered appearance of the synaptic ultrastructure in the hippocampus, the region of the brain involved in cognitive ability including memory.

Drug agency evaluations of lanthanum carbonate

Oral administration of lanthanum carbonate to rats of both sexes, at doses up to 2000 mg/kg bw/day, did not affect mating behaviour or fertility (US FDA, 2008).

Developmental toxicity was noted at high doses, at 2000 mg/kg bw/day in the rat and 1500 mg/kg bw/day in the rabbit. In the rat, delayed eye opening, reduction in body weight gain and delayed sexual development, manifesting as preputial separation for the males and vaginal opening for the females, were seen in the offspring. In the rabbit there was a reduction in maternal weight gain and food consumption, increased post-implantation loss, reduced foetal weights and delayed skeletal ossification (US FDA, 2008; Health Canada, 2005).

The reduced foetal weights were attributed to reduced phosphate absorption and/or stomach irritation in the dams and not as a direct effect of lanthanum exposure since there were no observed effects of placental transfer in the rats (Health Canada, 2005).

The Swedish (MPA) reported that learning, behaviour and reproductive performance of rat pups was assessed in an unspecified number of studies however experimental details and results were not provided (Swedish MPA, 2006). The Swedish MPA evaluation showed delayed postnatal development in the rat and rabbit, consistent with the findings of the US FDA and Health Canada.

Summary

The reproductive toxicity of orally administered lanthanum carbonate was investigated in studies covering the complete reproductive cycle (rat) and foetal development (rabbit). Based on the reported evaluation of these studies by the US FDA, the chemical did not produce any gross abnormalities or malformations, but, at high dose levels, some developmental effects were observed in the rat and rabbit.

Neurotoxicity of orally administered lanthanum chloride was assessed in four developmental studies in rodents. In the most reliable study, a LOAEL of 2 mg/kg bw/day is established based on neurobehavioural deficits manifested from impairment in spatial learning and memory abilities in rats, a reduction in brain cell numbers and effects on brain biochemistry. The NOAEL was 0.1 mg/kg bw/day. Further evidence of an effect of the chemical on neurobehavioural parameters was presented in two other studies, however these studies did not demonstrate clear dose-response relationships for several of the affected endpoints.

7.3 Effects observed in humans

No human data were submitted in the original assessment of Phoslock™. Published literature on bentonite clay and lanthanum chloride was used as surrogate data for assessment of effects of Phoslock™ in humans. The original assessment concluded that inhalation of dusts of dried bentonite clay (or dried Phoslock™) may cause chronic lung disease such as bronchial asthma due to the crystalline silica content, although the current Phoslock™ formulation no longer contains silica. The limited epidemiological data for rare earth compounds indicated that they can contribute to the risk of pneumoconiosis and chronic pulmonary reactions, however no evidence of similar effects was observed with lanthanum chloride in workers (NICNAS, 2001).

For the secondary notification assessment of Phoslock™, new studies in humans are available for lanthanum carbonate from clinical investigations on Fosrenol® for therapeutic use. Epidemiological studies investigating environmental lanthanide exposure and neurotoxicity are also available.

7.3.1 Clinical studies (ND)

Repeated dose toxicity

In a randomised, multicentre study of cognitive function in Stage 5 chronic kidney disease patients on haemodialysis (Altmann et al., 2007), 360 patients were divided into two groups receiving lanthanum carbonate versus standard therapy. In the lanthanum group, males (n = 104) and females (n = 75) of mean age and weight 54.4 years and 80.2 kg, respectively, received a titrated oral dose ranging from 375 to 3000 mg lanthanum carbonate per day for up to 2 years. The standard therapy group was maintained on their pre-study phosphate binder. Cognitive function was assessed at 6 monthly intervals using computer controlled tasks from the Cognitive Drug Research (CDR) cognitive assessment system that assesses attention, working memory and reaction time. The full 2 years of treatment and assessment was completed by 47/174 lanthanum patients and 77/178 standard therapy patients with the

difference in withdrawal rates explained by a protocol option for the latter to allow a change in or addition of another approved therapy and still remain in the study. Although a decline in cognitive function from baseline over the 2-year period was observed in both groups, the deterioration was similar in both groups and not exacerbated by lanthanum treatment. As the doses administered during the trial were variable, this study is of limited value in determining a NOAEL.

In a combined safety and efficacy study (Finn, 2006), 390 male and 292 female, mixed-race haemodialysis patients with end-stage renal disease (ESRD) (mean age and weight 53.8 years and 80.6 kg, respectively) received a titrated oral dose of either 375, 750, 1500, 2250 or 3000 mg lanthanum carbonate per day for up to 2 years. A matched control group of 415 males and 262 females were maintained on a (predominantly calcium-based) standard therapy. Pre- and follow-up checks comprised of physical examination, vital signs, clinical chemistry, bone biopsy, ECG and mini-mental state examination. The full 2 years of treatment was completed by 196/682 lanthanum patients and 321/677 standard therapy patients with the difference in withdrawal rates explained by a protocol option for the latter allowing a switch or addition of therapy and thus a greater likelihood to continue treatment. No clinically significant differences were observed among the treatment groups with regard to changes in any of the parameters examined. Although a greater incidence of nausea and vomiting was observed in the lanthanum group, the authors attributed this to patient non-compliance and concluded that lanthanum carbonate was well tolerated in this study. However, due to the variation in the dose administered during the trial, this study is of limited value in determining a robust NOAEL and consequently the results are not presented here in detail.

Additional clinical studies in ESRD patients (no normal controls included) are available that investigated standard drug safety parameters including vital signs, ECG, clinical chemistry, haematology and bone biochemistry (D'Haese et al., 2003; Hutchison et al., 2006; Joy et al., 2003; Spasovski et al., 2006) following use of lanthanum carbonate for up to 3 years. Adverse effects described in these trials were generally considered to be minor and were associated with gastrointestinal upset (e.g. abdominal pain, nausea and vomiting). No significant increase in overall frequency of lanthanum related-effects compared with other standard therapies was seen in these parameters. This study is of limited value and therefore not discussed further.

Drug agency evaluations of lanthanum carbonate

The US FDA (2008), Swedish MPA (2006) and Health Canada (2007) evaluated the clinical studies described above and considered that the most frequently reported adverse events were gastrointestinal in nature and did not present any major concerns. However, one agency did note that potential effects associated with the long term deposition of lanthanum in bone, liver and the gastrointestinal tract were not examined in these studies (Health Canada, 2007).

7.3.2 Epidemiological studies (ND)

No epidemiological studies of workers or populations specifically exposed to lanthanum are available. Several studies have attempted to relate health outcomes with environmental exposure to some rare earth elements (REEs), including lanthanum which is classed as a light rare earth element.

Published epidemiological surveys have reported that REEs could affect the cognitive functions of children living in the REE enriched regions of rural China as described in the studies below.

Zhu et al. (1996) found that children aged from 6 to 9 years from two REE-rich areas have a statistically significant younger mental age and lower intelligence quotients, measured by the Hiskey-Nebraska Test of Learning Aptitude, compared with control children living in a REE normal region.

Similar results were obtained in a more recent study (Fan et al., 2004) using the 'Drawing a Man' test where children aged 7-10 years living near a REE ore area had significantly lower intelligence quotients than those living in regions with normal REE levels. Blood levels of each of the 15 REEs measured in the children were, on average, 1.73 times higher in the exposure group than the control group, with lanthanum, cerium and yttrium the most elevated at 2.26, 1.88 and 2 times higher, respectively. These results were reported in an abstract translated in English from the full Chinese study and methodological details were not fully described making it difficult to assess confounding factors in the study.

Another study examined the linkage between changes in the bioelectrical activity of the central nervous system and exposure to REEs. Somatosensory evoked potential (SEP) tests conducted on 21 adult villagers from a high REE county in China showed statistically significant shortening of nerve conduction from the thalamus to the primary somatosensory responsive region, suggesting to the authors that chronic exposure to REEs may cause subclinical damage to the brain (Zhu et al., 1997).

All of the studies were poorly documented and inconclusive for determination of the hazards of lanthanum due to their inherent limitations such as the absence of direct exposure measurements and the potential of confounding factors, such as co-exposure to other chemicals in the environment. Accordingly, it is not possible to attribute the observed effects to any specific chemical.

7.4 Regulatory classifications based on hazard

In Australia, determination of whether a substance is hazardous to health of workers is based on the *Approved Criteria for Classifying Hazardous Substances* (NOHSC, 2004), which covers physicochemical properties, toxicological and ecotoxicological effects.

The classification for health effects is based on experimental studies. The hazard classifications based on the overall data including new data provided for secondary classification are presented below.

7.4.1 Physicochemical hazards

Based on the known physicochemical properties, Phoslock™ is not considered to be dangerous due to its physicochemical properties.

7.4.2 Health hazards

Assessment and classification as a new chemical (NA/899)

The new chemicals assessment NA/899 determined Phoslock™ to be non-hazardous based on the data provided for the two constituent chemicals, lanthanum chloride and bentonite. It was not classified as a hazardous substance in accordance with the *Approved Criteria for Classifying Hazardous Substances* (NOHSC, 2004), thus not listed in the Hazardous Substances Information System (http://hsis.ascc.gov.au/SearchHS.aspx).

7.4.3 Classification based on assessment of new data

The health effects of Phoslock™ are predominantly associated with the toxicity of its ionic lanthanum component. Thus, the classification of Phoslock™ will be based on the assessment of toxicological data, where available, for lanthanum carbonate, lanthanum chloride, and lanthanum nitrate.

The classification of Phoslock™ from the secondary notification assessment is based on the Third revised edition of the *Globally Harmonized System of Classification and Labelling of Chemicals* (GHS) (United Nations, 2009) as specified in the *Model Work Health and Safety Regulations* (Safe Work Australia, 2011c) and the *Approved Criteria for Classifying Hazardous Substances* (NOHSC, 2004).

Acute toxicity

No new acute toxicity studies were available for lanthanum carbonate, lanthanum chloride, and lanthanum nitrate.

Classification: Based on the available data from the original notification, Phoslock™ is not classified as hazardous for acute toxicity under the GHS (United Nations, 2009) and the *Approved Criteria for Classifying Hazardous Substances* (NOHSC, 2004).

Repeated dose toxicity

Significant toxic effects are related to repeated exposures to lanthanum carbonate, lanthanum chloride and lanthanum nitrate; the target organs include the stomach (stomach inflammation and hyperplasia), liver (histological changes and hepatotoxicity), and brain (changes in neurotransmitter levels).

In the critical study, male rats gavaged lanthanum chloride daily for 5 months showed a marked decrease in the Ca^{2+}-ATPase activity observed at the top dose of 40 mg/kg bw/day. There were significant reductions in the concentrations of the brain neurotransmitters dopamine, dihydroxyphenylacetic acid and noradrenaline at this dose and decreased level of 5-hydroxytryptamine at 2 mg/kg bw/day. Although a NOAEL of 0.1 mg/kg bw/day was assigned, clear toxicological significance of the effects of lanthanum chloride below 40 mg/kg bw/day is not well established. These changes are considered to be minor variations in clinical biochemistry and have uncertain or minimal toxicological significance.

Classification: Based on the available repeated dose toxicity studies for lanthanum carbonate, lanthanum chloride and lanthanum nitrate, Phoslock™ does not meet the criteria under the GHS (United Nations, 2009) for specific target organ toxicity following repeated exposure and the *Approved Criteria for Classifying Hazardous Substances* (NOHSC, 2004) for repeated dose or sub chronic toxicity.

Genotoxicity

Lanthanum carbonate was negative in a point mutation assay in bacteria (with and without metabolic activation) up to a cytotoxic concentration. In mammalian cells in vitro, conflicting results were observed. In a gene mutation study, a negative result was reported. However, in a chromosome aberration study, an equivocal result was reported (+/- S9) with the increase in aberration frequency within the historical control range and apparent at marginally cytotoxic concentrations that were at the solubility limit.

In somatic cells in vivo, negative results were seen in the two available bone marrow chromosome aberration studies using lanthanum carbonate and lanthanum chloride and a UDS study in rat liver in the presence of lanthanum chloride. Overall, there are a number of well conducted in vitro and in vivo studies available and no reliable evidence of a genotoxic potential.

Classification: Based on the available in vitro and animal data for lanthanum carbonate and lanthanum chloride, Phoslock™ does not meet the criteria under the GHS (United Nations, 2009) and the *Approved Criteria for Classifying Hazardous Substances* (NOHSC, 2004) for classification as a mutagen.

Carcinogenicity

Treatment of rats and mice for up to 104 weeks with lanthanum carbonate at doses up to 1500 mg/kg bw/day showed an increased incidence of glandular stomach adenomas in male mice at 1500 mg/kg bw/day. Pathological evaluation concluded that they were likely associated with an age related exacerbation of spontaneous pathological stomach changes in CD-1 mice. Increased incidence of malignant tumours was not observed in the rats.

Classification: Based on the available animal data for lanthanum carbonate, Phoslock™ does not meet the criteria under the GHS (United Nations, 2009) and the *Approved Criteria for Classifying Hazardous Substances* (NOHSC, 2004) for classification as a carcinogen.

Reproductive and developmental toxicity

The reproductive toxicity of orally administered lanthanum carbonate was investigated in studies covering the complete reproductive cycle (rat) and foetal development (rabbit) from drug agency evaluations of lanthanum carbonate. The chemical did not produce any gross abnormalities or malformations, but some reproductive toxicity at high dose levels (up to 2000 mg/kg bw/day in rat and up to 1500 mg/kg bw/day in rabbit) were observed with increased post-implantation losses and delayed skeletal ossification in the rat and delayed post-natal development in the rabbit. In the rabbit there were reduced foetal weights, which were attributed to reduced phosphate absorption and/or stomach irritation in the dams and not as a direct effect of lanthanum exposure. Limited reporting of treatment effects on dams was presented in these studies.

Manifestations of developmental toxicity at high doses, at 2000 mg/kg bw/day in the rat and 1500 mg/kg bw/day in the rabbit, were also observed. In the rat, delayed sexual development, manifesting as preputial separation for the males and vaginal opening for the females, were seen in the offspring.

The doses used in the drug agency evaluations of lanthanum carbonate were above the doses at which effects (i.e. reduced body weight gain, stomach and liver effects) were seen in repeated dose studies. This indicates that lanthanum carbonate is not specifically toxic to reproduction.

Four rodent developmental studies were evaluated for neurodevelopmental effects of lanthanum chloride. Clear dose-response relationships for several of the affected endpoints were not established in two studies. In the critical study, a LOAEL of 2 mg/kg bw/day is established based on neurobehavioural deficits from learning decrements correlated with brain oxidative stress, a reduction in brain cell numbers and effects on brain biochemistry. The NOAEL was 0.1 mg/kg bw/day. Maternal toxicity was not evaluated in all these studies.

Available epidemiological surveys attempted to relate health effects from environmental exposure to rare earth elements (REE), including lanthanum which is classified as a light REE. The poorly documented surveys were inconclusive of the determination of lanthanum hazards since the studies did not include direct exposure measurements and there could be a possibility of confounding factors (e.g. co-exposure to other chemicals in the environment) which make it difficult to attach an effect to any specific chemical.

The basis of classification of substances toxic to reproduction according to the *Approved Criteria for Classifying Hazardous Substances* (NOHSC, 2004) looks into effects on male and female fertility, and developmental toxicity. Classification of reproductive toxicity under the GHS (United Nations, 2009) considers adverse effects on sexual function and fertility, and adverse effects on development of the offspring.

Developmental effects should give due consideration to the possible influence of maternal toxicity (NOHSC, 2004; United Nations, 2009). Developmental effects at the administered doses from the neurotoxicity tests could not be analysed for any potential influence from treatment-related effects to dams since only limited maternal toxicity studies were presented. The neurodevelopmental findings in a study including direct dosing to pups post-weaning are related to repeat dose neurological findings in adults. The close relationship between the findings in the repeat dose and neurodevelopmental studies suggest that the findings are not specific developmental effects. The differences in NOAEL may result from heightened sensitivity of the juveniles compared with adults.

Classification: Based on the total weight of evidence evaluating the reproductive and developmental toxicity studies on lanthanum chloride and lanthanum carbonate, and published epidemiological surveys regarding rare earth elements (including lanthanum which is classed as a light rare earth element), Phoslock™ does not meet the criteria under the GHS (United Nations, 2009) and the *Approved Criteria for Classifying Hazardous Substances* (NOHSC, 2004) for classification as a reproductive toxicant.

8. Environmental Hazard Assessment

8.1 Introduction

All laboratory effects studies submitted as part of the original assessment of Phoslock™ were focussed on representative species from three trophic levels of aquatic ecosystems (algae, zooplankton, and fish). Apart from some monitoring of sediment microbial mass and bottom dwelling fish in field trials, no studies were presented on the effects of this chemical on sediment dwelling species, such as burrowing amphipods. This was considered a significant deficiency given that a key functional aspect of Phoslock™ is the formation of a layer of lanthanum-rich clay particles on the surface of sediments that is intended to intercept the flux of phosphorus from the sediment compartment to the water column, which promotes algal blooms (PWS, 2008d).

The original assessment of Phoslock™ found that the chemical was potentially toxic to aquatic biota, especially freshwater zooplankton, with a chronic (7-day) EC50 value of 0.43 mg/L in artificial softwater (NICNAS, 2001). It was further concluded that the ecotoxicity of Phoslock™ is principally due to lanthanum ions that leach from the chemical in water. This conclusion was supported by a study which showed that filtered leachates of Phoslock™ containing significant concentrations of dissolved lanthanum were very toxic to the water flea, *Ceriodaphnia dubia*, in synthetic soft water (48-hour EC50 = 80 µg/L) (NICNAS, 2001); and a contemporaneous study by Barry & Meehan (2000) which showed that lanthanum was very toxic to another species of water flea, *Daphnia carinata*, in soft water (48-hour EC50 = 43 µg/L). The results of toxicity tests on fish and algae were somewhat inconclusive in the tests available at the time. As a result, consumer communities represented by zooplankton were considered most likely to be adversely affected by dissolved lanthanum, and hence Phoslock™. The 48-hour EC50 for dissolved lanthanum in synthetic soft water leachates of Phoslock™ (80 µg/L) was therefore taken to be indicative of the concentration of dissolved lanthanum that could have adverse effects on aquatic ecosystems (NICNAS, 2001).

For the Secondary Notification assessment, several published studies on the effects of Phoslock™ and lanthanum to aquatic organisms have been assessed and presented in the succeeding sections.

8.2 Effects on aquatic organisms (ND)

8.2.1 Algae

The Hazardous Substances and New Organisms (HSNO) of New Zealand requires the determination of ecotoxic thresholds with a base set of toxicity tests for fish (acute and chronic), daphnia (acute and chronic), and algae. In the HSNO determination for Phoslock™, the "Eureka 1" granular formulation was utilised in all the tests. The Toxicity Characteristic Leaching Procedure (TCLP) method of the US EPA was employed wherein 50 g/L of Phoslock™ granules was combined with synthetic soft water (phosphorus-free, hardness 32 mg/L $CaCO_3$). Martin & Hickey (2004) stated that water samples were analysed for total and dissolved lanthanum using a boiling nitric acid digestion procedure and ICP-MS.

In the algal growth test of *Pseudokirchneriella subcapitata*, the microplate method was utilised in accordance with reference procedures of the US EPA and Environment Canada. Phoslock™ test solutions were prepared after passing the mixture of the granules (50 g/L) and synthetic soft water (phosphorus-free, hardness 32 mg/L $CaCO_3$) through a membrane filter (0.45 µm). The acute 72-hour EC50 for cell growth and the NOEC were 15000 and 6250 mg/L Phoslock™, respectively (Martin & Hickey, 2004).

8.2.2 Cladocera

In the Martin & Hickey (2004) study described above, the Phoslock™ leachate test solutions for laboratory-cultured *Daphnia magna* tests were prepared using the *Daphnia* culture water adjusted to a hardness of 40-50 mg/L $CaCO_3$. The authors conducted the *Daphnia* test in accordance with the OECD acute immobilisation test method. A 37% mortality was reported for the highest test concentration (50 g/L) at the end of testing. The acute 48-hour EC50 for immobility was >50000 mg/L Phoslock™ and the NOEC was 25000 mg/L Phoslock™.

Lürling & Tolman (2010) investigated the effects of lanthanum and Phoslock™ on the growth and survival of *D. magna* obtained from Lake Zwemlust in Netherlands and maintained in the laboratory. All the animals were given *Scenedesmus obliquus* (10 mm^3/L) as food. The 5-day growth and survival effects of Phoslock™ on the *D. magna* were analysed for the following levels: 0, 5, 50, 100, 500, and 5000 mg/L Phoslock™. No deaths were recorded up to the 100 mg/L treatment, 89% survival in the 500 mg/L dose and all the animals died in the 5000 mg/L dose. For weight-based growths, the EC50 and NOEC were 871 and 100 mg/L Phoslock™, respectively. For the length-based growths, the EC50 and NOEC were 1557 and 500 mg/L Phoslock™, respectively (Lürling & Tolman, 2010). The authors speculated that the observed effects were possibly due to feeding inhibition by suspended clay particles, rather than direct toxicity of bioavailable lanthanum.

The effects of Phoslock™ on aquatic organisms were examined by Watson-Leung (2009) before the product's field application in the Lake Simcoe watershed in Canada. Laboratory toxicity testing was conducted on the following in-house cultured organisms: freshwater cladoceran *Daphnia magna*, midge *Chironomus dilutus*, amphipod *Hyazella azteca*, mayfly nymph *Hexagenia* spp, and rainbow trout Oncorhynchus mykiss. All the water and sediment samples were analysed for total nitrogen, total and soluble (FRP) phosphorus, pH, alkalinity, and total and soluble lanthanum.

For the *D. magna* tests, the maximum stock concentration used was 6.8 g/L Phoslock™, wherein the granules were ground and mixed with either dechlorinated tap water or Scanlon Pond water. In 200 mL of test solution, 12 neonates were placed, starved for 48 h, and observed for mortality, mobility and other effects. The author stated that the standard Environment Canada methodology for determining acute lethality of effluents to *D. magna* was utilised. The calculated LC50 for the Phoslock™ solution with dechlorinated Toronto tap water (hardness 128 mg/L $CaCO_3$) was 4.9 g/L Phoslock™. For the Scanlon Pond solution, 42% mortality was observed at the highest dose, hence LC50 was given as >6.8 g/L Phoslock™. The author indicated that mortalities seen in Daphnia appeared to be caused by physical entrapment in Phoslock™ and some organisms physically trapped in the Phoslock™ slurry were alive upon test termination. In the dilution series conducted for the Phoslock™ test concentrations used in this study, the total and soluble lanthanum was lower in the 6.8 g/L dose (194.4 and 14 mg/L, respectively) than in the 3.4 g/L dose (239.4 and 63.3 mg/L, respectively) (Watson-Leung, 2009).

8.2.3 Fish

In the study of Martin & Hickey (2004), Phoslock™ test solutions were prepared using dechlorinated Hamilton City Council tap water (30-40 mg/L $CaCO_3$). *O. mykiss* fish fry were subjected to the OECD fish acute toxicity test. The toxicity to the fish fry appeared in the first 48 hours of the test and no mortality was observed after this period. The acute 96-hour EC50 for survival and the NOEC were 4350 and <3125 mg/L Phoslock™, respectively. The TCLP solution was filtered using a 40 μm nylon mesh.

For the rainbow trout *O. mykiss* tested in the study by Watson-Leung (2009), the highest level used was 13.6 g/L Phoslock™ prepared from ground granules mixed with dechlorinated Toronto tap water (hardness 128 mg/L $CaCO_3$). Two litres of stock solution per gram of juvenile rainbow trout were utilised (10 fish each Phoslock™ concentration) and the unfed fish monitored for 96 hours for mortality, mobility and other effects. The author stated that the standard Environment Canada methodology for determining acute lethality of effluents to rainbow trout was employed. No mortality was observed at any of the Phoslock™ concentrations tested. The 96-hour LC50 determined for this test was >13.6 g/L Phoslock™.

The 28-day LC50 for lanthanum to rainbow trout eggs in a rapid toxicity screening study has been reported as 20 µg/L (Birge et al., 1978, 1979, 1980), and presumably is less toxic over 4 days.

In the acute toxicity study of Clearwater & Hickey (2004), *O. mykiss* was treated (up to 400 mg/L Phoslock™) and toxicity was observed in accordance with the Environment Canada method for determining acute lethality. Test vessels contained a layer of sand in the base. Phoslock™ granules were pre-weighed and sprinkled evenly over the water surface. Fish mortality increased from 0 to up to 50% (although highly variable ±45%) between the 40 and 200 mg/L Phoslock™ treatments, but dissolved lanthanum measured at the end of the test in these treatments was only 10 and 14 µg/L, respectively. 100% mortality was observed at the highest dose with total lanthanum level of 88 µg/L (dissolved lanthanum not reported at this treatment). The 4-day rainbow trout LC50 and NOEC were 200 and 40 mg/L Phoslock™, respectively. Authors of the study commented that the data suggest either dissolved lanthanum is not primarily responsible for increased trout mortality or the dose response curve for dissolved lanthanum is very steep because mortality rates changed markedly with only very small increases in dissolved lanthanum (~4 µg/L). Additionally, the pH in the final solution was reported to be as high as 10; this is likely to be an additional stressor. Toxicity due to aluminium dissolving from the sediment is also likely at this pH.

The sub-acute toxicity of dissolved lanthanum from Phoslock™ or soluble lanthanum salts to other fish species seems to be much lower with 96-hour E(L)C50s ranging from >127 µg/L in rainbow fish (Stauber 2000) to 23,000 µg/L in zebrafish (RIVM 2000). The shortcomings demonstrated in the rainbow trout data, which indicate that apparent toxicity varies by orders of magnitude depending on water chemistry are also expected to apply in these studies. It is not possible to clearly identify sensitive species from these results.

8.2.4 Sediment-dwelling organisms

Tests with sediment present were undertaken by Clearwater & Hickey (2004) with sediment-dwelling species comprising of the midge *Polypedilum parvidum*, amphipod *Phreatogammarus helmsii*, and worm *Lumbriculus variegatus* using the 10-day amphipod chronic sediment toxicity methodology of the National Institute of Water and Atmospheric Research (NIWA) in the testing. The application rate utilised in the experiments followed the 200:1 ratio of Phoslock™ to phosphorus (assuming a baseline phosphorus level of 0.2 mg/L) with Phoslock™ concentrations up to 100 mg/L. The initial water hardness in the samples was >70 mg/L $CaCO_3$.

The mortality of *P. helmsii* increased from 20 to 70% in Phoslock™ doses of 20-40 mg/L with corresponding dissolved lanthanum levels of 6-8 µg/L measured at the end of the test. Authors of this study suggested that the mortality observed may be due to physical effects rather than direct toxicity of bioavailable lanthanum. The 10-day survival LC50 and NOEC

for *P. helmsii* were 33 and <20 mg/L Phoslock™, respectively. No changes in survival of *L. variegatus* (LC50 and NOEC = >1000 and 1000 mg/L) and *P. parvidum* (LC50 and NOEC = >400 and 400 mg/L) were observed for dissolved lanthanum levels up to 483 and 140 µg/L, respectively (Clearwater & Hickey, 2004).

In a similar 200:1 ratio (Phoslock™ to phosphorus) application rate, Clearwater (2004) conducted a 38-day toxicity test with sediment on the midge *Chironimus zealandicus* in accordance with the standard OECD procedure on chironomid toxicity testing. Phoslock™ level up to 400 mg/L was employed in the study that tested survival and emergence of the midge larvae. At the end of the test duration, dissolved lanthanum was 7.7 µg/L in the highest treatment. No toxicity was observed throughout the test so the 38-day LC50 and NOEC were >400 and 400 mg/L Phoslock™, respectively (for both survival and emergence).

Scanlon Pond water was collected and analysed for FRP to estimate the amount of Phoslock™ required for a 250:1 dosing ratio of Phoslock™ to phosphorus used in the study by Watson-Leung (2009). Ten litres of sediment from three sites in the Scanlon Pond reservoir (ecosystems part of Lake Simcoe), as well as control sediment, were collected and used for the testing. Five treatment regimes were developed for this study as well as two controls (sediment with laboratory dilution water or Scanlon Pond water). The survival and growth of three sediment-dwelling organisms (*C. dilutus*, *H. azteca*, and *Hexagenia* spp) were monitored in the sediment samples. For each organism, a 1:4 ratio of sediment to water was prepared and aerated overnight, after which the organisms were placed in the mixture. The author stated that the Environment Canada methodologies on whole-sediment tests for survival and growth were used.

There were no significant differences between the treatments and controls on the growth rate and survival (>80% survival in all the samples) of the mayfly *Hexagenia* spp. and the LC50 for the 21-day survival and growth test was >450 mg/L Phoslock™. For the *H. azteca* test, all the amphipod treatments met the growth and survival test acceptability criteria and the LC50 for the 14-day survival and growth test was >3400 mg/L Phoslock™. No significant changes in treatments on the growth and survival of *C. dilutus* were observed and the LC50 for the 10-day survival and growth test was >3400 mg/L Phoslock™ (Watson-Leung, 2009).

8.2.5 Phoslock™ toxicity with phosphorus/phosphate addition

In the investigation by Martin & Hickey (2004), water samples from the fish toxicity tests were collected, filtered (40 µm) and analysed for ionic lanthanum as part of the fish mortality mitigation by addition of phosphorus, wherein 100% Phoslock™ elutriate (50 g/L) was treated with different phosphorus doses (20, 100, 500, 2500 µg/L). Control treatment consisted of no Phoslock™ elutriate and no phosphorus. The pH of the treatments was modified to 7.5 by adding 2480 mg/L $NaHCO_3$ solution. At 50 g/L Phoslock™, 100 and 0% mortality were observed for the 0 and 2500 µg/L phosphorus treatments, respectively, after 72 hours of exposure. The levels of dissolved lanthanum significantly decreased in the 2500 µg/L phosphorus dose compared with the 500 µg/L dose.

Contradictory to this result is the study by Lürling & Tolman (2010) which examined the effects of lanthanum with and without phosphate (330 µg/L) on the 14-day growth of *Daphnia magna* for 10 different lanthanum treatments (0, 33, 100, 330, and 1000 µg/L La in nanopure water). The animals treated with lanthanum and phosphate showed significant dose-dependent growth reductions. *D. magna* lengths were significantly shorter in the 100, 330, and 1000 µg/L La in the treatments with phosphate compared with the phosphate-free

samples. The lanthanum treatments with phosphate presented marked precipitation of the *S. obliquus* which affected the apparent toxicity of lanthanum. The authors stated that probably in the highest lanthanum dose with phosphate treatment, the precipitation of the suspended food particles may have contributed to the toxicity of the *D. magna* suggesting starvation effects rather than lanthanum toxicity.

8.3 Monitoring of aquatic organisms from Phoslock™ field applications (ND)

Toxicity testing was undertaken using the water flea *Ceriodaphnia dubia* in freshwaters from Fitzroy Falls in NSW to which slurries of Phoslock™ granules had been added (ESA, 2008). This used the so-called China formulation. No toxicity was seen in either 48-hour acute toxicity tests with up to 50 mg/L Phoslock™ or a 7-day chronic reproductive impairment test with up to 1 mg/L Phoslock™. The 48-hour test was a static non-renewal test, while in the 7-day test, water was renewed daily. Measured dissolved lanthanum concentrations in the 48-hour test ranged from 15 µg/L for a 0.75 mg/L Phoslock™ application, to 480 µg/L for 50 mg/L Phoslock™. It was argued that some of this high concentration could have been filterable but particulate (or colloidal). The highest dissolved lanthanum level in the chronic test was 20 µg/L for a 1 mg/L Phoslock™ application.

The Ecotoxicology Laboratory of the NSW Department of Environment and Climate Change (DECC) in 2008 undertook 48-hour *Ceriodaphnia* tests on a sample of Phoslock™ granules used in Deep Creek Reservoir (Section 6.2.5) and a sample of Phoslock™ granules intended to be used in Fitzroy Falls reservoir (alkalinity 20 mg/L $CaCO_3$). The diluent for both tests was water taken from Fitzroy Falls reservoir (NSW DECC, 2008b). Similar results were obtained for both samples, with an EC50 value in the range 150-160 µg/L La, and with immobilisation occurring at 170 µg/L La with a LOEC of 20 mg/L Phoslock™. This is inconsistent with the ESA results on the Fitzroy Falls water which showed no toxicity at 330-480 µg La/L (50 mg/L Phoslock™).

Field biological monitoring study was conducted at Torrens Lake as part of the overall monitoring program associated with the March 2007 Phoslock™ application (Section 6.2.5). Macroinvertebrates and zooplanktons were collected by shore-based dip netting from a 10 m edge of habitat over a 100 m section of stream which was defined as a sampling site. Samples were collected from an upstream control site and from three sites within the lake on the day of application (6 March) and 8 days later (14 March). The identity of species and their approximate abundance in netted samples were based on visual observations by a scientist knowledgeable about the local macroinvertebrate fauna (SA EPA, 2007). The survey revealed abundant taxa throughout the lake including zooplanktons (Ostracoda and Cycopoida), midge larvae from the family Chironomidae, baetid mayflies, water boatman and the mosquito fish. Although there were some site variations in the faunal assemblages, the authors concluded that there were no major changes in the community structure for the resident macroinvertebrates and zooplankton species related to the application of Phoslock™. There were no obvious impacts on the resident birds, fish and frog populations of the lake, and no mortalities among any of these groups noted during the survey period.

The biological monitoring study also included follow on laboratory tests of the effects of Phoslock™ on the moulting and emergence into adulthood of mayfly nymphs from the family Baetidae (*Cloeon fluviatile*) resident in the River Torrens. The test samples included water from an upstream control site, water from the treated area of the lake collected before and after application of Phoslock™, a slurry of Phoslock™ granules in Torrens Lake water collected at the point of application, and dilutions of the slurry in control water. The mortality and developmental success of the mayflies were monitored over 100 hours in open topped vials each containing a single mayfly nymph (5 replicates per sample). The

chemistry of the water samples used to incubate the nymphs, including soluble and total lanthanum concentrations, were not reported. However, this report indicated that Torrens Lake has both hard water and high dissolved organic carbon concentrations (SA EPA, 2007).

A total of 4 nymphs died during the trial including 2 (out of 5) from an undiluted sample of Phoslock™ slurry collected from the boat during application of the chemical to the lake. The study author noted that the dead nymphs were replaced 18 hours after test initiation and that the replacements were alive at the end of the test. It was also noted that in test samples containing slurries, live mayflies were found to be sitting on a layer of settled slurry particles overlain by clear water at the end of the study period. These findings were taken to indicate there were no significant acute toxic responses of mayfly nymphs to Phoslock™ under the conditions of this test (SA EPA, 2007).

The developmental effects of Phoslock™ on mayfly nymphs were more difficult to assess as nymphs were able to moult and emerge as adults from all samples containing slurries of Phoslock™, however no emergence of adults was noted in the control group. The study author speculated that Phoslock™ may have been responsible for a subtle increase in the rate of development of the nymphs, but overall the results were inconclusive with respect to the possible chronic toxicity of Phoslock™ to this aquatic invertebrate (SA EPA, 2007).

In parallel with the application of Phoslock™ (as Bentophos®) to the Silbersee and Bärensee lakes in Germany described in Section 6.2.6, laboratory toxicity analyses on *Daphnia magna* and fish egg (species not specified) were conducted by the Institut Dr Nowak (IDN). No study details were presented, however, the authors indicated that the tests were performed in accordance with the German standard methods for the tolerance of *Daphnia* to the toxicity of waste water by dilution series, as well as the determination of the effect of non-acute toxicity of wastewater on fish egg development. The reported EC50 values for *D. magna* and fish egg toxicity were 103 and 150 mg/L La^{3+}, respectively (IDN, 2008a and 2008b).

As part of the application monitoring regime in Lake Rauwbraken in Netherlands (Section 6.2.6), lake water samples were collected and the health of *Daphnia galeata*, a zooplankton species which commonly resides in the lake, was determined by measurements of both algal grazing and survival (van Oosterhout & Lürling, 2011). Before Phoslock™ application, the zooplankton was harvested by filtering the water through a net (100 μm). Harvesting was done on the following occasions: April-June 2006 and 2007, and January-April 2008.

Water from the lake was filtered (0.45 μm) and same-sized *D. galeata* were submerged. This served as the stock culture for the grazing experiment and the green algae *Scenedesmus obliquus* was used as food for the zooplankton. Similar dosages (9 and 67 mg/L) used in the Phoslock™ field application of the lake and an additional dosage (670 mg/L) were employed in the study. After an incubation period, the clearance rates were calculated based on the levels of chlorophyll-a measured in the solution for the different treatment types. In the survival experiment, *D. galeata* were placed in 800 mL filtered (0.45 μm) lake water with the following treatments: no food, 2 μg/L chlorophyll-a *S. obliquus* food, and 20 μg/L chlorophyll-a *S. obliquus* food. The food concentrations were adjusted to the desired level before analysis. The samples were checked for *D. galeata* survival on days 1, 4, 8, 9 to 15.

In the grazing study, the clearance rate was estimated based on the decrease in chlorophyll-a concentration in the Phoslock™-treated samples relative to controls. The clearance rates calculated in the grazing study showed a significant decrease as the Phoslock™ dosing was increased. In the survival study, the highest survival rate (75%) occurred in the animals given the highest food concentration. The death rates were significantly different between the two food treatments (2 and 20 μg/L chlorophyll-a *S. obliquus*). However, the findings of

these tests were not consistent with the field measurements wherein the authors claimed that *D. galeata* juveniles increased in numbers during and after Phoslock™ application in Lake Rauwbraken (van Oosterhout & Lürling, 2011).

Fish health monitoring of rainbow trout, *O. mykiss*, koura *Paranephrops planifrons*, and bully *Gobiomorphus cotidianus* on Lake Okareka was performed by Landman et al. (2007) prior to and after the third Phoslock™ application to the lake on 20-21 March 2007 (Section 6.2.6). The fish health monitoring was conducted on pre-application in March, April, and June 2007, with nearby Lake Tikitapu used as reference lake. Following necropsy, haematology, histology, and tissue metals analysis were conducted on the organs. Apart from the significant differences in the female trout's liver and spleen somatic indices between the two lakes, the general physiological parameters remained the same throughout the monitoring period. The authors stated that the variations and inconsistencies in the haematological findings of the koura and bully blood samples could be possibly due to seasonal changes, temporal differences, or fish size-related effects. Significant increase in the levels of lanthanum in the liver of *O. mykiss* sampled in Lake Okareka was observed compared with the reference lake trout samples. Male trout showed a more rapid accumulation of lanthanum compared with the females. The toxic effects of the accumulated lanthanum in the liver were not analysed further (Landman et al., 2007).

8.4 Summary of effects to aquatic organisms

A summary of the studies on Phoslock™ effects on aquatic organisms discussed in Section 8.2 and in the Appendix is presented in Table 8.1. The toxicity tests largely targeted the impact of particular dose rates of Phoslock™ or lanthanum leachates/solutions based on laboratory jar tests. The measured levels of dissolved lanthanum at the E/LC50 are also presented for each study when values were available. The tests on sediment-dwelling species were conducted in the presence of sediments simulating the actual field situation whereas tests on the other organisms were mostly water-only (no sediments) tests.

Toxic effects associated with Phoslock™ in laboratory experiments are most likely due to bioavailable lanthanum released to the overlying water as the surface applied granules or granule slurry settle through the water column to the sediment, although in some tests, toxicity may have been due to physical effects of Phoslock™ or food limitation. However, it is important to note that the dissolved lanthanum analysed does not necessarily correspond to the ionic lanthanum found in solution. In the case of benthic organisms, exposure will also include sediment porewater and ingestion of sediment particles. The relationship between dose and response for both ionic lanthanum and Phoslock™ is not obvious, especially in sediment porewaters. Experimental artefacts, such as a loss of alkalinity due to precipitation of carbonate as insoluble lanthanum salts, and food reduction due to precipitation may complicate the conduct of ecotoxicological testing of this chemical in the laboratory. As can be seen from the EC50 or LC50 values in Table 8.1, huge variations in toxicity were observed even in tests on single species, and these could be dependent on parameters (e.g. pH, water hardness) other than the levels of dissolved lanthanum available for uptake. Water hardness from the various water media could have contributed to the toxicity differences as well as other aspects of the methodology used in the studies.

The studies of Stauber (2000), Martin & Hickey (2004), and ESA (2008) utilised Phoslock™ leachates extracted using the US EPA TCLP method compared with the other studies wherein Phoslock™ or lanthanum chloride solutions were utilised in the laboratory tests. The TCLP method eliminates the effects of high particulate solids on aquatic organisms. However, if the mechanism of Phoslock™ is to settle through the water column through flocculation of lanthanum phosphate and/or lanthanum carbonate from surface water applications, then the toxicity results from leachates by the TCLP method may not be useful in interpreting the effects of the chemical to the aquatic species. Toxicity results from

a methodology that mimics the mechanism of Phoslock™ in the actual field applications may be more appropriate in determining the effects to the various aquatic organisms.

In the cladoceran acute toxicity studies, lanthanum chloride effects decreased with increasing water hardness (filtered tap water to ASTM hard water) as demonstrated by Barry & Meehan (2000) noting that lanthanum was rapidly lost from solution at concentrations of less than 1000 µg/L with the loss amounting to 20-60% of the nominal concentration. Stauber & Binet (2000) showed much lesser acute effects to lanthanum chloride in synthetic soft water containing 48 mg/L $NaHCO_3$. This comparison is limited by the different species (*D. carinata* and *C. dubia*) of the zooplankton for which the sensitivity to the chemical may vary. The acute toxicity studies (ESA, 2008 and NSW DECC, 2008) involving *C. dubia* dosed with the more recent China granular formulation of Phoslock™ showed comparable results which gave lower EC50 values than the *D. magna* study by Watson-Leung (2009) using the same formulation in high alkalinity pond waters. The TCLP leachate studies by Martin & Hickey (2004) and Stauber (2000) exhibited similar toxic effects even though the authors used slightly dissimilar sized filters (40 µm vs. 0.45 µm).

The chronic cladoceran tests presented large variations in EC50 values considering the different endpoints observed in the investigations. Phoslock™ toxicity was examined for growth (weight-based and length-based) in *D. magna* (Lürling & Tolman, 2010), immobilisation and reproduction of *C. dubia* (ESA, 2008), and reproduction in *C. dubia* (Stauber, 2000). These studies used different Phoslock™ formulations as well as testing methodologies and showed wide-ranging toxic effects. However, the study by Lürling & Tolman (2010) used an artificial medium containing the metal chelator ethylenediaminetetraacetic acid (EDTA) which could have affected the effects of Phoslock™ in the cladoceran by the complexation of La^{3+}.

Table 8.1. Summary of ecotoxicity testing results with E/LC50 endpoints.

Species	Duration, day	Treatment	Hardness (alkalinity), mg/L $CaCO_3$	E/LC50	Measured dissolved La, µg/L, at E/LC50	Study
Algae						
Pseudokirchneriella subcapitata	3	Phoslock (Eureka)	32	**15000** mg/L Phoslock	NR	Martin & Hickey, 2004
P. subcapitata	3	$LaCl_3$ solution	NR	**450** µg/L La (nominal)	10	Stauber & Binet, 2000
Cladocera						
Daphnia carinata	2	$LaCl_3$ solution	22	**43** (~100 nominal) µg/L La	**43**	Barry & Meehan, 2000
			98	**49** (~1000 nominal) µg/L La	**49**	
			160	**1180** (~6000 nominal) µg/L La	**1180**	
D. magna	2	Phoslock (Eureka)	40-50	**>50000** mg/L Phoslock	NR	Martin & Hickey, 2004
D. magna	2	Phoslock (China)	128	**4900** mg/L Phoslock	NR	Watson-Leung, 2009
			(203)	**>6800** mg/L Phoslock		
D. magna	5	Phoslock (China)	88	**871** WG, **1557** LG mg/L Phoslock (nominal)	NR	Lürling & Tolman, 2010
D. magna	14	$La(NO_3)_3$ solution	NR	**>1000** µg/L La (nominal)	**100** (in 330 nominal)	Lürling & Tolman, 2010
D. magna	NR	Phoslock (China)	NR	**103** mg/L La	NR	IDN, 2008a & 2008b
Ceriodaphnia dubia	2	Phoslock (Eureka)	40-48 (30-35)	**24500** mg/L Phoslock	**80**	Stauber, 2000
C. dubia	2	$LaCl_3$ solution	NR	**5000** µg/L La (nominal)	NR	Stauber & Binet, 2000
C. dubia	2	Phoslock (China)	(13)	**>50** mg/L Phoslock	**330**	ESA, 2008
C. dubia	2	Phoslock (China)	NR	**22-26** mg/L Phoslock (nominal)	**150-160**	NSW DECC, 2008
C. dubia	7	Phoslock (Eureka)	40-48 (30-35)	**20500** mg/L Phoslock	**820**	Stauber, 2000
C. dubia	7	$LaCl_3$ solution	NR	**430** µg/L La (nominal)	NR	Stauber & Binet, 2000
C. dubia	7	Phoslock (China)	(20)	**>1** mg/L Phoslock	**20**	ESA, 2008

Table 8.1. cont.

Species	Duration, day	Treatment	Hardness (alkalinity), mg/L $CaCO_3$	E/LC50	Measured dissolved La, µg/L, at E/LC50	Study
Fish						
Melanotaenia duboulayi	4	Phoslock (Eureka)	40-48 (30-35)	**>50000** mg/L Phoslock	**127**	Stauber, 2000
M. duboulayi	4	$LaCl_3$ solution	NR	**<600** µg/L La (nominal)	**<600**	Stauber & Binet, 2000
M. splendida	4	Phoslock VDM4	NR	**>50000** mg/L Phoslock	NR	ESA, 2008
		Phoslock (Eureka)	NR	**>50000** mg/L Phoslock	NR	
Oncorhynchus mykiss	2	Phoslock (Eureka)	30-40	**4350** mg/L Phoslock	NR	Martin & Hickey, 2004
O. mykiss	4	Phoslock (China)	128	**>13600** mg/L Phoslock	NR	Watson-Leung, 2009
O. mykiss	4	Phoslock (Eureka)	33-51	**200** mg/L Phoslock	**14**	Clearwater & Hickey, 2004
O. mykiss	28	NR	NR	**20** µg/L La	NR	Birge et al., 1978, 1979, 1980
Sediment-dwelling species						
Hyalella azteca	14	Phoslock (China)	(Up to 208)	**>3400** mg/L Phoslock	NR	Watson-Leung, 2009
Hexagenia spp.	21	Phoslock (China)	(Up to 105)	**>450** mg/L Phoslock	NR	Watson-Leung, 2009
Chironomus dilutus	10	Phoslock (China)	(Up to 179)	**>3400** mg/L Phoslock	**880** (total)	Watson-Leung, 2009
C. zealandicus	38	Phoslock (Eureka)	123-202	**>400** mg/L Phoslock	**7.7**	Clearwater, 2004
Polypedilum parvidum	10	Phoslock (Eureka)	>70	**>400** mg/L Phoslock	**140**	Clearwater & Hickey, 2004
Phreatogammarus helmsii	10	Phoslock (Eureka)	>70	**33** mg/L Phoslock	**6 - 8**	Clearwater & Hickey, 2004
Lumbriculus variegatus	10	Phoslock (Eureka)	>70	**>1000** mg/L Phoslock	**483**	Clearwater & Hickey, 2004

NR = not reported; ~ = approximately; La = lanthanum; $LaCl_3$ = lanthanum chloride solution; WG = weight-based growth; LG = length-based growth

The fish studies by Stauber (2000), Clearwater & Hickey (2004), and Watson-Leung (2009) tested 4-day sub-acute toxicity in juvenile fish (*M. duboulayi* and *O. mykiss*) compared with the other studies that looked at fish fry/larvae sensitivity or immobilisation. Clearwater & Hickey (2004) had the lowest EC50, however, the authors suggested that toxicity to Phoslock™ may not be the only factor attributed to mortality due to the minimal dissolved lanthanum increase measured in the treatments. The TCLP leachate studies of Stauber (2000) and ESA (2008) on the rainbowfish *Melanotaenia* have the same EC50 for two different species, although description of the latter study was not presented in full. The toxicity of *O. mykiss* to Phoslock™ TCLP leachate was greater possibly due to the more sensitive nature of the *Oncorhynchus* than the *Melanotaenia* (Martin & Hickey, 2004).

All the chronic toxicity studies performed on the sediment-dwelling organisms were done with sediments using natural lake/pond/spring water. The tests were all conducted under high alkaline and/or hard waters. The amphipod *P. helmsii* was the most sensitive sediment species when treated with the earlier Phoslock™ formulation.

In the identification of the appropriate studies to take forward in this assessment, the studies that determined the effects of Phoslock™ leachates by the TCLP method will not be included. The toxic effects that will simulate the actual settling mechanism of Phoslock™ in the water column and eventually in the sediment through flocculation will be considered. The studies that did not utilise TCLP were conducted with standard protocols of the OECD, Environment Canada and/or US EPA.

The disparity between the lanthanum levels in treatment of Phoslock™ or lanthanum solutions is apparent in the studies that have measured levels of dissolved lanthanum presented in Table 8.1. The likelihood that parameters other than direct toxicity of lanthanum present in Phoslock™ gave rise to the toxic effects has been reported in some of the studies. It is important to recognise that the Phoslock™ dosing rates used in the investigations are higher than the actual field application rates (Sections 6.2.5 and 6.2.6) presented in Table 8.2.

Table 8.2. Nominal dose rates from Phoslock™ field applications.

Water body	Nominal dose (mg/L)
Deep Creek Reservoir	12
Bärensee	73.7
Zwemplas De Kuil	149.3
Het Groene Eiland	107.7
Clatto Reservoir	68.6

The most sensitive species in the toxicity tests is the invertebrate *P. helmsii* with chronic EC50 and NOEC of 33 and <20 mg/L Phoslock™, respectively. At these doses, the concentrations of dissolved lanthanum measured at the end of the test were approximately 8 (or lower) and 6 µg/L, respectively. In the Deep Creek Reservoir, the nominal dose was 12 mg/L and the measured dissolved lanthanum was 102 µg/L; this can be compared with an acute EC50 value of 22-26 mg/L Phoslock™ obtained in a *C. dubia* test with measured dissolved lanthanum level of 150-160 µg/L. These discrepancies demonstrate that nominal Phoslock™ is not always a good measure of applied dose since differences in dissolved lanthanum levels have been observed for different water conditions.

8.5 Predicted no-effect concentration (PNEC)

A PNEC for lanthanum was not calculated for the original assessment of Phoslock™. Instead, a concentration of 20 µg/L dissolved lanthanum was taken to provide an acceptably low threshold concentration for acute toxic effects on sensitive invertebrates in soft water (NICNAS, 2001). This threshold concentration has since been used as a limit concentration on dissolved lanthanum in a number of field applications of Phoslock™ to water bodies in Australia (e.g. Water Corp. WA, 2008).

The *Australian and New Zealand Guidelines for Fresh and Marine Water Quality* ('Water Quality Guidelines': ANZECC/ARMCANZ, 2000a) presented guidance for the derivation of guideline trigger values for toxicants in aquatic ecosystems. The guideline impact concentrations are taken as the unacceptable contaminant levels in the aquatic ecosystem. The determination of these values is classified into high, moderate or low reliability based on the availability and quality of toxicity tests. The trigger values can be taken as comparable to PNECs and has been adapted in this assessment.

An environmental concern level (ECL) for lanthanum was derived in the Water Quality Guidelines for which the ECLs are only intended to serve as indicative interim working levels for toxicants in aquatic ecosystems. Interim levels are assigned as low reliability guideline trigger value. The ECL is defined as the concentration of a chemical which may cause adverse environmental effects and is considered equivalent to a PNEC (OECD, 1995). The current freshwater ECL for lanthanum in Australia is 0.04 µg/L, extrapolated from the lowest 48-hour EC50 for *D. carinata* in soft dechlorinated Melbourne tap water (43 µg/L) as measured by Barry and Meehan (2000).

The extrapolation was based on the assessment factor method, which in this case involved applying the typical maximum assessment factor (AF) of 1000 to the lowest measured acute EC50 for lanthanum (ECL = EC50/AF). The maximum assessment factor was applied in this case as the ECL was extrapolated from a median lethal effect concentration for the acute effects of ionic lanthanum on a single aquatic species (OECD, 1995). The ECL for lanthanum is likely to be conservative since it was extrapolated from acute effects on a single aquatic species. It is recognised that for situations where the concentration of a toxicant exceeds the ECL, water quality managers may need to refine the estimated concentration at which the toxicant will not cause a significant adverse effect on an ecosystem.

The Water Quality Guidelines provided guidance in deriving guideline trigger values, and the minimum data requirements that must be satisfied for high, moderate, and low reliability which have been compiled and presented in Table 8.3. For metallic toxicants, it should also be possible to calibrate measured end points for water chemistry effects.

The derivation of a revised PNEC for ionic lanthanum that can be used for aquatic risk assessments and which takes account of the influence of environmental water chemistry on the effects of this toxicant is needed for the improved risk characterisation and risk management of Phoslock™. The approach recommended in the Water Quality Guidelines is considered appropriate for this purpose as the method is consistent with approaches to the effects assessment of metallic toxicants in aquatic ecosystems.

Table 8.3. Type of toxicity tests, data requirements, and assessment factors used to derive the trigger values described in ANZECC/ARMCANZ (2000a).

Trigger Value Reliability	Minimum data requirements	Assessment Factor
High	Chronic NOEC toxicity data for five different species (1 fish, 2 invertebrates of different taxonomic groups, 1 alga or plant, 1 from any of these taxonomic groups or from a different taxonomic group)	10
Moderate	Acute E/LC50 toxicity data for five different species (1 fish, 2 invertebrates of different taxonomic groups, 1 alga or plant, 1 from any of these taxonomic groups or a different taxonomic group)	10 x ACR or 100
Low	Acute and/or chronic E/LC50 for three species (1 fish, 1 invertebrate, and 1 algae)	10 x ACR or 100
	Chronic NOEC toxicity data for three species (1 fish, 1 invertebrate, and 1 algae)	20
	Any toxicity data	1000

ACR = acute-to-chronic ratio if chronic test available for same species in same test, else a default ACR of 10.

Based on the hierarchical framework of the Water Quality Guidelines (Table 8.3) and the currently available lanthanum toxicity studies on aquatic organisms (Table 8.1), the trigger value for lanthanum falls under the low reliability grade, satisfying the minimum data requirements of acute and/or chronic E/LC50 for three species (fish, invertebrate, and algae). The dissolved lanthanum concentrations at the E/LC50 were measured on the premise that the ionic lanthanum is in equilibrium with the Phoslock™ or lanthanum chloride solutions. A revised low reliability trigger value for freshwater can then be estimated for lanthanum from the available acute toxicity data, and the lowest EC50 as indicated below:

- Algae: 3d EC50 – 450 μg/L nominal La (10 μg/L dissolved La)
- Invertebrate: 10d EC50 – 33 mg/L Phoslock™ (6-8 μg/L dissolved La)
- Fish: 4d EC50 – 200 mg/L Phoslock™ (14 μg/L dissolved La)

Based on the lowest dissolved lanthanum level of 6 μg/L in equilibrium with an acute Phoslock™ EC50 of 33 mg/L, a freshwater low reliability trigger value for ionic lanthanum was derived using the assessment factor of 100 and could be revised to 0.06 μg/L. This value is considered comparable to a PNEC in this assessment, but it has not been calibrated for the influence of water chemistry.

9. Human Health Risk Characterisation

9.1 Methodology

A margin of exposure (MOE) methodology is used frequently in international assessments to characterise risks to human health associated with exposure to chemicals (EC, 2003). The risk characterisation is conducted by comparing quantitative information on exposure to the NOAEL and deriving a margin of exposure as follows:

1. Identification of critical health effect(s).
2. Identification of the most appropriate/reliable NOAEL (if available) for the critical effect(s).
3. Where appropriate, comparison of the estimated or measured human dose or exposure (EHD) to provide an MOE:

$$MOE = NOAEL/EHD$$

4. Characterisation of risk, by evaluating whether the MOE indicates a concern for the human population under consideration.

The MOE provides a measure of the likelihood that a particular adverse health effect will occur under the conditions of exposure. As the MOE increases, the risk of potential adverse effects decreases. In deciding whether the MOE is of sufficient magnitude, expert judgment is required. Such judgments are usually made on a case-by-case basis, and should take into account uncertainties arising in the risk assessment process such as the completeness and quality of the database, the nature and severity of effect(s) and intra/inter species variability.

A MOE of 100 or greater is usually not regarded as an indication of concern as it covers the conservative default uncertainty factors of 10 each for intraspecies and interspecies variability used for risk characterisation.

In this assessment, the MOE methodology was used for characterising the occupational and public health risks from Phoslock™ exposure.

9.2 Critical health effects

Adverse effects to human health are characterised in detail in Section 7.

Animal and human studies have shown that uptake of lanthanum, after ingestion of lanthanum carbonate, is extremely low and is eliminated predominantly via non-renal mechanisms. A proportion of the bioavailable lanthanum is deposited primarily in the liver and bone. There is some evidence from animal studies that lanthanum crosses the blood-brain barrier.

For lanthanum compounds, the limited repeat dose oral studies available were of narrow focus and not compliant with OECD test guidelines. A NOAEL of 1000 mg/kg bw/day was identified in rats from a 12-week bone toxicity study of lanthanum carbonate while in a 6-month study of lanthanum chloride a NOAEL of 2 mg/kg bw/day was determined based on changes in elemental distribution in the brain and a reduction in the concentration of neurotransmitter levels in the brain. While these changes were not correlated with microscopic adverse effects, they do correlate with the neurodevelopmental effects seen in the pups treated with similar doses from weaning to adulthood (below).

In a 6-month study of lanthanum nitrate, a NOAEL of 10 mg/kg bw/day was identified based on histological changes to the liver (including cellular infiltration) at the top dose although, as data were not reported, it was not obvious whether this effect was dose related. In contrast, no adverse histologic changes were seen in the livers of uraemic rats administered lanthanum carbonate at an estimated lanthanum dose of 1660 mg/kg bw/day

for 4 weeks and normal and uraemic rats at an estimated dose of 1134 mg/kg bw/day over a 110 day period.

Although human data are available from long-term, clinical studies in renal patients using lanthanum carbonate, they provide no information on the dose-response relationship to allow the identification of a robust NOAEL or profile the systemic toxicity of lanthanum.

Developmental toxicity is investigated by exposing pregnant dams or neonatal animals and looking for adverse effects in the development of the offspring. Neurodevelopmental toxicity is studied to examine changes in behaviour of the offspring from exposure of the dam during pregnancy and lactation. Behavioural changes may be due to toxic effects on the central nervous system and/or other organs.

With respect to developmental studies, only animal data are available for lanthanum chloride using the oral route of exposure. In the mouse, no maternal effects were seen up to and including the top dose of 120 mg/kg bw/day however no reliable data were available to identify a robust NOAEL in the progeny. Delay in emergence of swimming behaviour in the mouse offspring was observed with a NOAEL of 11 mg/kg bw/day. In a neurobehavioural study in rats, impairment in spatial learning and memory abilities were manifested in rats at up to 2 mg/kg bw/day with a NOAEL established to be 0.1 mg/kg bw/day. These effects were seen in pups treated with the relevant doses from weaning to adulthood.

The critical rodent studies of up to six months lanthanum chloride exposure are shown in Table 9.1.

Table 9.1. Critical studies for NOAEL determination.

Toxicity observed	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Effect at LOAEL	Reference
Brain biochemistry	2	40	Decreased neurotransmitter levels dopamine and noradrenaline and alterations in brain elemental distribution	Feng et al., 2006b
Neurobehaviour	11	23	Delay in swimming behaviour	Briner et al., 2000
Neurobehaviour	0.1	2	Learning decrements correlated with brain oxidative stress	He et al., 2008

The studies in Table 9.1 support a NOAEL for brain biochemistry and neurodevelopmental effects from lanthanum chloride in the dose range of 0.1-11 mg/kg bw/day. Within this range, the appropriate NOAELs for risk estimation are: 2 mg/kg bw/day for adults from brain biochemistry changes in adult rats (Feng et al., 2006b) and 0.1 mg/kg bw/day for children from neurobehavioural changes observed in rat pups (He et al., 2008).

The NOAELs to be used for risk estimation are adjusted according to the ionic lanthanum content of lanthanum chloride but not accounting for hydration of water (MW La / MW LaCl3 = 0.566). The MOE was calculated based on the NOAEL of 0.06 mg/kg bw/day for children and 1.2 mg/kg bw/day for adults and the estimated received exposures for children and adults shown in Table 5.2.

9.3 Occupational health risk estimation

Phoslock™ is imported in sealed bags within shipping containers and occupational exposure to Phoslock™ during transport is low due to the handling of the bags. Similarly, current distribution, warehouse and retail workers will also have low exposure as they will only

handle sealed products. Repackaging of powder was historically a manual process but is not forecast to take place in future. Risk of adverse effects during transport and warehouse storage of Phoslock™ is minimal as potential exposure to the chemical is unlikely except in cases of accidental spills.

During debagging, transfer of granules via conveyer belt to a barge, slurrying and application of Phoslock™, inhalation and dermal contact are the most likely routes of exposure for field workers. Ocular exposure may also occur due to accidental splashes. Occupational risk due to these routes of exposure in the field is estimated to be low and can be reduced by wearing the appropriate personal protective equipment including facemask, gloves, protective clothing and eyewear when handling the chemical. Exposure to dust and aerosols during large scale operations will also be limited by the benefit of forced draught ventilation as a result of the movement of the application barge on the water surface.

Under normal occupational conditions, the risk to workers of adverse health effects from Phoslock™ is low.

9.4 Public health risk estimation

Products containing the chemical are not available to the public. In the event of the product being available to the public for domestic use, the potential for exposure during do it yourself (DIY) application of Phoslock™ carries the same exposure risk as for the field workers in Section 9.3 but the duration of contact would be shorter and use would be less frequent. However, the retail consumer would be less likely to wear the appropriate protective equipment so that the risk of exposure to dust, spills and splashes is increased.

Although, the Phoslock™ clay is highly water insoluble, there is potential for leaching of lanthanum ions from the clay which could lead to elevated soluble lanthanum levels in the water. It is possible that drinking water containing soluble lanthanum will be available in the public domain and consequently there is potential for widespread public exposure.

Based on human health considerations and concentrations that are as low as reasonably practicable to achieve, the draft *Australian Drinking Water Guidelines* (ADWG) proposed in 2010 that the concentration of lanthanum in drinking water as supplied to the consumer should not exceed 0.002 mg/L (NHMRC, 2010). The studies used for critical health endpoints in the derivation of this concentration is substantially the same as those identified as critical hazard studies in this report (Section 9.2). NHMRC does not include this concentration in the current ADWG (NHMRC, 2011). The WHO *Guidelines for Drinking Water Quality* (2006) provide no guideline values for lanthanum or other lanthanides and it is not listed as a contaminant in the US EPA's *National Primary Drinking Water Regulations* (US EPA, 2007).

Netherlands' National Institute of Public Health and the Environment (RIVM) have derived a Maximum Permissible Concentration (MPC) of 10.1 µg/L for lanthanum in fresh surface water in Netherlands based on a Maximum Permissible Addition (MPA) of 10 µg/L and a background concentration of 0.1 µg/L (Sneller et al., 2000).

A risk assessment was carried out by TERA (1999) to develop a reference dose for lanthanum chloride and lanthanum oxide. The reference dose (RfD) is an estimate of a continuous exposure to humans (including sensitive sub-groups) that is likely to be without an appreciable risk of deleterious effects during a lifetime. For lanthanum chloride and lanthanum oxide, RfDs of 0.005 and 0.02 mg/kg bw/day (based on NOAELs of 15 and 52 mg/kg bw/day, respectively) were determined using an uncertainty factor of 3000. However, due to the limited nature of the data available at the time, confidence in the estimated RfDs for both salts was considered to be medium to low.

Estimation of margin of exposure

Risk estimates take into account the likelihood of neurodevelopmental effects related to long term exposure through ingestion of drinking water, possibly containing soluble lanthanum, from Phoslock™.

Dissociated lanthanum can be absorbed into the body after ingestion of drinking water which is the primary route of systemic uptake. The ingestion exposure of soluble lanthanum was determined using controlled and reasonable worst-case concentrations of 2 and 33 µg/L, respectively, of lanthanum in drinking water as supplied to the consumer. The MOE was calculated based on the NOAEL of 0.06 mg/kg bw/day for children and 1.2 mg/kg bw/day for adults and the estimated received exposures for children and adults shown in Table 5.2. The estimated MOE is presented in Table 9.2.

Table 9.2. Risk parameters and calculated MOE from estimated ingestion exposure to drinking water of adults and children.

	NOAEL	D_{oral} (µg/kg bw/day)		MOE	
	(mg/kg bw/day)	Controlled	Worst-case	Controlled	Worst-case
Child	0.06	0.2	3.3	**300**	18
Adult	1.2	0.06	0.94	**20000**	1277

The risk estimates for the controlled exposures of children and adults are above 100 and hence indicate low risk of adverse developmental effects. However, the MOE for the reasonable worst-case scenario is significantly less than 100 for children and marginal for adults.

The MOE for the controlled scenario assumes that a controlled concentration of lanthanum in drinking water (0.002 mg/L) is not exceeded in water supplied to households by water utilities. It should be noted that some environmental water bodies treated with Phoslock™ have levels that are significantly higher than this concentration, and this is taken into account by providing the reasonable worst-case MOE calculations.

Uncertainties in the risk estimate

Uncertainties in any risk characterisation process arise from inadequate information, assumptions made during the process, and variability in experimental conditions. The uncertainties in the characterisation of risk from Phoslock™ arise mainly from limited data and include:

- Absence of specific data for oral bioavailability of lanthanum from Phoslock™-;
- Absence of Australian or overseas comprehensive data on the lanthanum content in drinking water from water bodies treated with Phoslock™;
- Lack of data on the health effects of lanthanum released from Phoslock™ in young and/or adult humans following acute and/or repeated exposures;
- Removal of residual lanthanum from water by treatment processes not taken into account in the estimation; and

There is a high degree of uncertainty associated with the estimates of the reasonable worst-case exposures, as it is not known whether the concentration of soluble lanthanum in Phoslock™-treated waters eight weeks post-application is the actual concentration that will be present in the drinking water to be available to the Australian population. The assumptions used for the reasonable worst-case exposures may lead to overestimation of risk to the population.

Areas of concern

The sensitivity of individuals and subpopulations to the critical health effects associated with lanthanum exposure may vary significantly.

There are no known standard treatment protocols employed uniformly by the state and territory authorities. However, it is expected that lanthanum levels will be reduced by the processes used to prepare potable water fit for human consumption (e.g. coagulation, flocculation, sedimentation, filtration, pH correction, anti-scaling, or a combination of these) and adhering to the recommended maximum levels obtained by using the standardised methodology of the *Australian Drinking Water Guidelines*. Residues of compounds of lanthanum modified bentonite (Phoslock™), lanthanum phosphate ($LaPO_4$) and other particulate bound lanthanum which may form part of the total suspended solids load are expected to be removed during any of these processes. In addition, lanthanum levels in drinking water provided to the consumer need to be monitored and managed so as not to present risk of adverse developmental effects.

10. Environmental Risk Characterisation

10.1 Methodology

The process of comparing the predicted environmental concentration (PEC) and predicted no-effect concentration (PNEC) for quantitative risk characterisation is frequently utilised in assessments to characterise risks from chemicals discharged and/or released to the environment (EC, 2003). The ratio of the PEC and PNEC is also known as the risk quotient (RQ) where RQ = PEC/PNEC. The risk characterisation is conducted by comparing quantitative information on the environmental effects under conditions of exposure.

In deciding whether the RQ is of sufficient magnitude, expert judgment is required. Such judgments are usually made on a case-by-case basis, and should take into account uncertainties arising in the risk assessment process. The PEC/PNEC ratio or RQ aims to determine whether the risks are acceptable or unacceptable. A RQ of less than or equal to 1 is usually not regarded as an indication of environmental concern.

10.2 Risk estimates

The environmental effects of Phoslock™ are likely to be principally derived from the ionic (i.e. bioavailable) lanthanum released into environmental waters at varying quantities from field applications of the product. In addition, the environmental toxicity of ionic lanthanum is highly dependent on the water chemistry of the receiving water body. Currently, there are no satisfactory models to calculate either the PEC for ionic lanthanum derived from Phoslock™ or environmental end-points for bioavailable lanthanum species adjusted for the influence of water chemistry. It is therefore not possible to provide quantitative estimates of the environmental risks of Phoslock™ for all conceivable application scenarios. However, it is possible to provide an indication of the risks for a scenario based on the results of the field applications of Phoslock™ in various water bodies.

10.2.1 Water column

A risk quotient for application of Phoslock™ to a water body can be calculated if it is assumed that the PEC for dissolved lanthanum is comparable to the peak concentration of 220 µg/L as measured in the Deep Creek Reservoir and where the PNEC for lanthanum is 0.06 µg/L. Based on this approach, the exceptionally high RQ for Phoslock™ is 3667. This RQ is based on the dissolved lanthanum level that peaks 1-3 days after Phoslock™ application and a conservative low reliability PNEC; both factors could overestimate the risks to ionic lanthanum. If the highest leaching rate of 20 µg/L dissolved lanthanum in laboratory tests identified in the original assessment (NICNAS, 2001) was taken as the PEC, RQ is 333. The latter calculation indicates a potential risk of acute toxic effects on sensitive aquatic organisms from the use of Phoslock™ for the control of cyanobacteria blooms in water bodies with low water hardness.

At an exposure concentration of 220 µg/L of dissolved lanthanum in soft environmental water, most sensitive zooplankton would be expected to die within 48 hours based on the lowest measured EC50 of 43 µg La/L for water fleas in soft water from the study of Barry & Meehan (2000). While the concentration may be non-uniform, a concentration of this magnitude may be expected to show significant toxicity to sensitive species. This is consistent with environmental monitoring of Deep Creek Reservoir after application of Phoslock™ which showed almost no living zooplankton about 7 weeks following the trial. The loss of critical zooplankton populations could be expected to significantly disrupt food webs in functioning aquatic ecosystems, with flow on effects to other aquatic biota. Although the ionic lanthanum levels in treated water bodies would decrease over time as a

result of the formation of precipitates with oxyanions, especially phosphate and carbonate, and complexation with humic substances, the peak measurements of dissolved lanthanum from Deep Creek Reservoir after application of Phoslock™ suggest that potential environmentally toxic elevated levels of dissolved lanthanum can persist for several weeks after application of this chemical in soft water. The disruption to aquatic food webs through acute toxic effects on invertebrate populations could therefore be persistent and significant.

The possibilities for mitigating the environmental toxicity of Phoslock™ will be related, in the first instance, to reducing the total levels of lanthanum that are released into any treated water body and removing ionic lanthanum rapidly and irreversibly from the water column. An attempt to mitigate the effects of lanthanum release was made by Phoslock Water Solutions at Deep Creek Reservoir through the addition of 5 tonnes of soda ash to the water body, which was intended to aid the precipitation of any released ionic lanthanum as the insoluble salt, lanthanum carbonate (PWS, 2007b). This strategy did not work under field conditions possibly due to the close to 300:1 Phoslock™ to FRP application rate in this water body instead of the recommended dosing of 100:1.

Alternative strategies such as reduced application rates with appropriate intervals between applications to allow peak levels of dissolved lanthanum to reduce may be employed. The applicant has reported that other field applications that have done so did not demonstrate higher than expected levels of dissolved lanthanum.

While the application of Phoslock™ in Deep Creek Reservoir led to significant toxicity to aquatic species, there have been a range of other field uses of Phoslock™ in Australia and elsewhere without evident toxicity. A number of the other applications have involved similar or greater quantities of Phoslock™ relative to the water body volume, and several have also resulted in higher maximum dissolved lanthanum concentrations than those assumed in the original assessment. This indicates the complexity of the risk assessment as both the concentration of dissolved lanthanum and the toxicity of a given level of dissolved lanthanum appear to be very strongly affected by the water quality parameters for a given body of water. However, it should be noted that the application to Deep Creek Reservoir involved the greatest excess of Phoslock™ relative to FRP among all the field applications.

10.2.2 Sediment compartment

The sediment compartment of environmental water bodies treated with Phoslock™ will be the ultimate sink for the constituent clay particles of this chemical and the ionic lanthanum released from these particles. The formation of a capping layer at the sediment-water interface is stated to be a functional requirement for this chemical (PWS, 2008d). There is therefore considerable potential for accumulation of solids containing elevated concentrations of lanthanum in sediments. This aspect of the aquatic exposure of Phoslock™ will be especially significant for the long term risks to static or enclosed water bodies which may be subject to multiple treatments with this chemical.

The available data for the toxicity of Phoslock™ to benthic invertebrates do not indicate a high hazard to sediment-dwelling organisms, except for some amphipods. However, it is not clear whether the results of these studies provide for reliable indicators of long term hazard to benthic invertebrates, especially in the absence of measurements of porewater lanthanum concentrations and suitable exposure models for this compartment. Studies on the porewater concentration of lanthanum in sediments treated with Phoslock™ and additional sediment toxicity studies to establish the dose response to porewater lanthanum levels are essential to adequately address the impacts of Phoslock™ on the sediment compartment.

Applications where Phoslock™ is used in stoichiometric (or lower) concentrations relative to FRP are not expected to result in formation of a capping layer.

10.3 Site-specific risk assessment

The environmental risks of Phoslock™ are the environmental risks of the ionic lanthanum it contains. The environmental speciation, fate and toxicity of lanthanum are strongly dependent on water chemistry. It is also clear that the release of lanthanum from Phoslock™ is itself dependent on the chemistry of the environmental water. As the chemistry of water bodies is site-specific, it is reasonable to conclude that the environmental concentration, fate and toxicity of lanthanum derived from Phoslock™ will also be site-specific.

The concept of site-specific risk assessment is well established especially for metals in the environment (ANZECC/ARMCANZ, 2000a). This approach may be used to refine risk characterisations where generic models reveal a RQ greater than 1, as is the case for Phoslock™. More refined and specific RQs can be calculated when suitable models and data are available regarding the species sensitivities and exposure concentrations for a toxicant at individual sites.

10.3.1 Direct toxicity assessments

For situations where insufficient data or models are available, site-specific risk assessments could be based on direct toxicity assessments (DTA) of the toxicant in question conducted on appropriate test species in the environmental waters to be exposed (ANZECC/ARMCANZ, 2000a). The DTA approach to evaluating the site-specific risks of Phoslock™ is appropriate given the significant uncertainties in relation to the bioavailable concentration of ionic lanthanum that will be present in water bodies following treatment and the sensitivity of aquatic species exposed to these toxicants.

The principal aim of any direct toxicity studies is to ensure that biota in the water and sediment compartments of a receiving water body will not be adversely affected by Phoslock™ treatment. Some useful guidance on the principles and practice of DTA of environmental toxicants as applied to waters in Australia and New Zealand waters are described in the Water Quality Guidelines (ANZECC/ARMCANZ, 2000a). In the case of Phoslock™, the most relevant aspect of this guidance involves the assessment of the bioavailability and toxicity of lanthanum under site-specific conditions with sensitive species in local dilution waters. DTA could be aimed at refining the guideline trigger value to reflect site-specific biological sensitivities. To be effective, the subsequent application of Phoslock™ to any water body would have to include appropriate monitoring of bioavailable (i.e. ionic) lanthanum levels in the water and sediment compartments to ensure that levels of this toxicant remained below the revised local trigger value. If techniques for measurement of ionic or labile lanthanum are not readily available in the field, measuring dissolved lanthanum is likely to provide the closest surrogate for ionic lanthanum."

A simplified direct toxicity assessment procedure for Phoslock™ was recently used as part of the approvals process for the trial use of this chemical to control blue-green algal blooms in the Fitzroy Falls reservoir in NSW (ESA, 2008 & PWS, 2008b). This study involved the direct toxicity testing of Phoslock™ to C. dubia in environmental waters collected from the proposed application site at Fitzroy Falls at application rates spanning the intending field application rates, as discussed in Section 8.2. To be fully effective this approach would need to be expanded to cover more taxonomic groups including sensitive benthic invertebrates and, where possible, it would be desirable to include on-site microcosm or mesocosm testing as well. These studies should also include monitoring of total and bioavailable, or alternatively, dissolved lanthanum in water and sediment compartments before, during and after application of Phoslock™. In principle, well designed site-specific testing of the effects of Phoslock™ would provide more reliable predictions of the likely environmental impacts of the use of this chemical than any currently available predictive models.

The monitoring of aquatic organisms after Phoslock™ application is discussed in Section 8.3. The toxicity effects to the aquatic organisms from the monitoring studies are summarised in Table 10.1 and were observed from actual treatments of Phoslock™ and can be considered as post-application DTA.

Table 10.1. Ecotoxicity monitoring from Phoslock™-treated water bodies.

Species (Where)	Duration	Endpoint	Result (Reference)
Cloeon fluvitale (Torren Lake)	100 h	Mortality and development	Inconclusive chronic results; no significant acute toxic effects up to peak dissolved lanthanum of 100 μg/L (SA EPA, 2007)
Ceriodaphnia dubia (Fitzroy Falls)	2 d	EC50	150-160 μg/L dissolved lanthanum; immobilisation at 170 μg/L dissolved lanthanum (NSW DECC, 2008)
C. dubia (Fitzroy Falls)	2 d	EC50	330 μg/L dissolved lanthanum (ESA, 2008)
C. dubia (Fitzroy Falls)	7 d	Reproductive impairment	No toxicity observed up to the highest dissolved lanthanum level of 20 μg/L (ESA, 2008)
Oncorhynchus mykiss (Lake Okareka)	n.s.	Haematology, tissue analysis	Significant increase in liver lanthanum levels of Phoslock™-treated fish compared to reference fish (Landman et al., 2007)
Daphnia galeata (De Rauwbraken)	15 d	Grazing clearance rate	Significant decrease in clearance rates in Phoslock™ treatments compared to controls (van Oosterhout & Lürling, 2011)
D. magna (Silbersee)	n.s.	EC50	103 mg/L dissolved lanthanum (IDN, 2008a)

n.s. = not specified

10.3.2 Water chemistry parameters

The environmental risks associated with the use of Phoslock™ to prevent nuisance algal blooms in environmental water bodies have revealed a potentially high risk of adverse effects on aquatic organisms in some application scenarios. The risks are now recognised as highly site-specific due to the influence of water chemistry on the concentration and bioavailability of ionic lanthanum released from this chemical. Site-specific risk assessment procedures for Phoslock™ need to be developed. More detail in understanding the relationship between ionic lanthanum toxicity and pertinent water chemistry parameters is desirable.

pH

As discussed in Section 6.2.1, the phosphorus adsorption capacity of Phoslock™ in reverse osmosis water remained the same from pH 5-7 and decreased from pH 7-9 at Phoslock™ to FRP dosing ratio of 230:1. At an increased dosing ratio of up to 450:1, the reduction in adsorption capacity at pH >9 was still recorded (Ross et al., 2008). Similar results were reported by Ecowise (2004) wherein there were no effects observed on the FRP uptake of Phoslock™ in de-ionised water in pH changes from 5 to 9.

The decrease in phosphorus removal of Phoslock™ from pH 7-9 can be due to the reduction of binding sites on lanthanum from the formation of the hydroxyl compounds of lanthanum, with the precipitation of lanthanum hydroxides starting at pH 8.35 (Haghseresht et al., 2009). In alkaline natural waters, the presence of humic acids could also lower the adsorption capacity of Phoslock™ at this pH range.

Water hardness and alkalinity

The total concentration of calcium and magnesium ions is expressed as the concentration of calcium carbonate and is the measure of water hardness. Alkalinity, also given as calcium carbonate levels, is a measure of the buffering capacity (Health Canada, 1979) and is equal to the sum of the carbonate, bicarbonate and hydroxide content of water (NHMRC, 2011). In this assessment, the water hardness and alkalinity of an environmental water body are considered equivalent.

From the Phoslock™ and lanthanum solution toxicity results (Section 8) summarised in Tables 8.1 and 10.1, dissolved, and presumably ionic, lanthanum is less acutely toxic to aquatic invertebrates in hard waters (120 to <180 mg/L $CaCO_3$) than soft waters (0 to <60 mg/L $CaCO_3$), presumably due to the binding of the ionic lanthanum with carbonate ions in hard waters, as well as increased competition from multivalent hardness cations such as Ca^{2+} and Mg^{2+} for biotic ligands. However, these cations may also increase the release rate of La^{3+} from Phoslock™ by ion exchange in a more efficient manner than monovalent cations.

Within the cladoceran and fish investigations in soft waters, the studies conducted with the Phoslock™ leachates by the TCLP method showed lesser toxicity, which may vary in part due to the indirect effects of the leachates on the organisms tested, or potential indirect toxicity effects of Phoslock™ granules (e.g. physical entrapment or food limitation). The rainbow fish *Oncorhynchus mykiss* study in soft water (hardness 33-51 mg/L $CaCO_3$) had the lowest EC50 of 200 mg/L Phoslock™ containing a dissolved lanthanum level of 14 μg/L at the end of the test.

In the presence of high levels of $CaCO_3$ (123-202 mg/L) in a chronic (38-day) sediment toxicity test, the midge larvae *Chironomus zealandicus* has an EC50 of >400 mg/L Phoslock™ with a dissolved lanthanum level of 7.7 μg/L. The apparent lack of toxicity of the organisms may be attributed to the competition for lanthanum binding sites of the phosphate in the environmental water and sediment with the carbonate in the very hard water used in the chronic midge study. The hard waters and high dissolved organic carbon levels understood to be present in Torrens Lake may have had a significant role in reducing the effects of the dissolved lanthanum (up to 100 μg/L) released from Phoslock™ to the mayfly *Cloeon fluviatile*. In contrast, the water only lanthanum chloride acute (2-day) toxicity testing of the crustacean *Daphnia carinata* reported an EC50 of 1180 μg/L dissolved lanthanum for ASTM water (hardness 160 mg/L $CaCO_3$), where the La^{3+} could have interacted with Ca^{2+} in the ASTM hard water and reduced the toxicity of dissolved lanthanum to the *D. carinata*. In addition, the absence of adsorbing phases such as sediment may have played a role in increasing the toxicity of the ionic lanthanum.

Dissolved organic carbon (DOC), dissolved oxygen (DO), and chlorophyll-a

High level of dissolved organic carbon would reduce the availability of lanthanum to the biotic ligands of aquatic organisms through complexation and/or adsorption, thus further reducing its apparent toxicity under the prevailing water chemistry conditions of the environmental water body. Complexation of lanthanum would, however, increase the concentration of lanthanum found in the dissolved phase, albeit not as free La^{3+}.

Salinity

The strong influence of salinity on dissolved lanthanum concentrations released from Phoslock™ was discussed in Section 6.2.4 in various environmental water samples, wherein excess ionic lanthanum was released as the salinity of the solution is increased.

10.3.3 Limitations in Phoslock™ application

The optimal conditions for Phoslock™ application that would yield minimal releases of dissolved lanthanum in environmental waters need to be established and, once established, can be used in the development of minimum data requirements needed for a site-specific application of the chemical.

The limiting factors that dictate the release of excess lanthanum ions in environmental waters are the water chemistry and the toxicity effects of ionic lanthanum to aquatic organisms. The release of dissolved lanthanum that is bioavailable is exacerbated by a number of physical, chemical and biological water parameters.

From the analysis of the laboratory studies, results from pre- and post-application monitoring and toxicity testing, excess dissolved lanthanum and/or toxicity from dissolved lanthanum that is in ionic form is observed in Phoslock™ applications in the following environmental waters:

- Generally in soft water or low alkalinity waters (<60 mg/L $CaCO_3$).
- In soft waters and very low levels of FRP. Ionic lanthanum is more likely to remain in solution since there is a lack of anions to bind with. The binding of La^{3+} ions and precipitation of insoluble lanthanum compounds is desired to render the ions unavailable for uptake by aquatic organisms.
- Water bodies with high salinity waters, increasing release of lanthanum from Phoslock™.

The pH of the water is also a contributing factor in terms of the effectiveness of Phoslock™ to adsorb the FRP. Although the adsorption capacity of Phoslock™ is stable in pH ranges 6 to 9, it is noteworthy to take into account the formation of humic acids at pH 8.35 and above. Humic acids binding with lanthanum ions decrease the active binding sites available for FRP.

The incidence of algal blooms brings a decrease in DO since the algae use up the DO in the decomposition process. The application of Phoslock™ would result in using additional DO being used in decomposition of algae trapped during the flocculation process with the FRP in the water. Thus, the timing of the Phoslock™ application should, as far as practicable, be avoided with an existing algal bloom since it may put additional stress on the environmental waters limiting the availability of DO for use by other aquatic organisms.

Rain events could also influence the release of dissolved lanthanum as observed from the field applications wherein resurfacing of the sediments resulted in increased levels of the soluble lanthanum. Phoslock™ application could be timed to not coincide with significant rain events.

11. Risk Management

11.1 Previous recommendations

The original assessment of Phoslock™ (NICNAS, 2001) contained the following recommendations to minimise the occupational exposure to the chemical:

- Safety goggles should be selected and fitted in accordance with Australian Standard (AS) 1336 (Standards Australia, 1994) to comply with Australian/New Zealand Standard (AS/NZS) 1337 (Standards Australia/Standards New Zealand, 1992); industrial clothing should conform to the specifications detailed in AS 2919 and AS 3765.1 (Standards Australia, 1990); gloves should conform to AS/NZS 2161.2 (Standards Australia/Standards New Zealand, 1994a); for facemask should conform to AS/NZS 1716 (Standards Australia/Standards New Zealand, 1994b) or other internationally acceptable standards.
- Spillage of the notified chemical should be avoided. Spillages should be cleaned up promptly and put into containers for disposal.
- A copy of the MSDS should be easily accessible to employees.
- Employers should ensure that NOHSC Exposure Standards for all of the components in lanthanum modified clay and inspirable dusts are not exceeded in the workplace. Effective ventilation and enclosed transfer and mixing should be used.

If products containing the notified chemical are hazardous to health in accordance with the NOHSC Approved Criteria in Classifying Hazardous Substances (NOHSC, 1999), workplace practices and control procedures consistent with state and territory hazardous substances regulations must be in operation.

The assessment also contained the following recommendations to minimise the environmental effects of the chemical:

- The manufacturing processes for lanthanum modified clay should be optimised to reduce the amount of free lanthanum released on application to waterways.
- Future commercial development trials of the notified chemical to collect additional ecological test data as identified and to be provided under secondary notification requirements.
- State & territory environmental agencies and/or appropriate regulatory authorities responsible for water quality are to be informed prior to each use.
- Adequate measures should be taken to protect sensitive benthic species such as gobies.
- Application of the clay slurry to water bodies using spraying techniques should be performed in a responsible manner. In so far as practically possible precautions should be taken to prevent over spray onto river and dam banks so as to minimise disturbance to flora and communities of shore dwelling fauna.
- Final outcomes of the commercial development trials should include a set of guidelines relating to appropriate use of the notified chemical and approved methods of clay slurry application to water bodies.

The environment-specific recommendations were appropriate since Phoslock™ was still undergoing development at the time of the new chemical notification and the original environmental risk assessment indicated that there was not a large safety margin for the intended use of this chemical. Specific technical measures to minimise the potential for

adverse environmental effects of Phoslock™, by limiting releases of dissolved lanthanum and avoiding exposure of sensitive benthic biota, were based on the unique environmental hazard properties of the new chemical identified at the time. Recommendations designed to minimise environmental effects through risk management were included. The most important of these are to inform the responsible local authorities prior to each use of Phoslock™ and to develop a set of guidelines for appropriate field applications of this chemical.

11.2 Current control measures

Phoslock™ is not available to the public for domestic use.

The exclusive use of Phoslock™ is for the treatment of environmental water bodies susceptible to cyanobacterial blooms and possibly other water remediation applications. All product applications are conducted by personnel authorised by the applicant upon completion of a 'Job Safety Analysis/Safe Work Method Statement' (JSA). Further information on the JSA is discussed in Section 4.1.3. Furthermore, PWS has provided the company's updated (PWS, 2012) Standard Operating Procedures (SOPs) which must be followed in Phoslock™ field applications to large water bodies (> 2 hectares). The SOPs indicated the undertaking of a risk assessment prior to application. The risk assessment involves evaluation of historical data of a water body to be supplied by the client, and laboratory tests using field water samples and various Phoslock™ doses. For drinking water reservoirs, PWS provided two key assumptions in Phoslock™ application:

- The reservoir could be made offline for 5-7 days as a measure to ensure the total binding of lanthanum ions with phosphate ions and complete settling of the solids to the sediment phase; and
- Phoslock™ to not be applied near the intake towers or rising mains that supply the raw water to the treatment plant.

PWS provided a certificate of compliance from the Water Quality Association, an international trade association representing the water treatment industry, which was issued in 11 November 2011 and expired on 31 December 2011 and the toxicological evaluation which is the basis of the certification. From the evaluation of Phoslock™-extracted lanthanum, the total allowable concentration (TAC) and single product allowable concentration (SPAC) values of 4 mg/L and 0.4 mg/L, respectively, were derived from toxicological studies using lanthanum carbonate as surrogate. However, the evaluation did not include neurodevelopmental studies on lanthanum.

The manufacture, use and disposal of industrial chemicals such as Phoslock™ are regulated by relevant state and territory authorities in Australia. In the state of NSW, the use of Phoslock™ for water remediation purposes requires a licence under the *Protection of the Environment Operations Act 1997* where it would result in pollution of waters as defined by the POEO Act (NSW OOW, 2009). Water pollution includes but is not limited to the addition of inorganic matter, liquid that contains suspended or dissolved solids, and chemical toxicants for which guidelines are prescribed in the *Australian and New Zealand Guidelines for Fresh and Marine Water Quality* ('Water Quality Guidelines': ANZECC/ARMCANZ, 2000a). They may include a variety of conditions such as concentration limits on added chemicals (limit conditions) and requirements for monitoring of chemical levels and biological health in the treated water body before and after application (monitoring and recording conditions). There are also a variety of reporting conditions including a requirement to report any "incidents causing or threatening material harm to the environment as soon as practicable" which can be used to alert water managers to adverse changes in the affected water body.

A licence was granted by the NSW Environment Protection Authority (NSW EPA) for the use of algicide or granular "Eureka 1" formulation of Phoslock™ as a short-term response to an algal emergency in Deep Creek Reservoir (Environment Protection Licence Number: 12279). A second licence has also been granted by the NSW EPA for a short term trial use of Phoslock™ to contain free reactive phosphorus in the waters of Fitzroy Falls (Environment Protection Licence Number: 12944). For the trial at Fitzroy Falls, a range of specific conditions were included. These include requirements to conduct both biological and chemical monitoring of the water body. A limit of 20 µg/L was placed on the allowable concentration of dissolved lanthanum. The dissolved lanthanum concentration limit was apparently based on the threshold concentration for acute aquatic toxic effects of dissolved lanthanum that was derived in the new chemical assessment of Phoslock™ (NICNAS, 2001).

In the licence for the use of Phoslock™ in Deep Creek Reservoir, general monitoring and reporting conditions that are on all licences do appear to have been effective in alerting the then NSW Government Department of Environment and Climate Change (DECC) that a pollution incident had occurred and triggering subsequent investigations including biological testing and measurements of dissolved lanthanum.

The SA EPA indicated that it currently licenses the application of Phoslock™ to Torrens Lake by the Adelaide City Council under Schedule 1 Part A (section 8(7)) of the Environment Protection Act 1993. The licence time is limited and has some site-specific conditions such as hardness trigger value and limitations to prevent application on days when rain is forecast. The SA Department of Health would need to be notified for any potential application of Phoslock™ to drinking water reservoirs in the state.

The licensing approach utilised by NSW and SA for the use of Phoslock™ provides an example of one mechanism for controlling the potential for adverse environmental effects of this chemical on water bodies with functioning ecosystems. However, in order for licensing or other approval mechanisms to be effective, it must be possible to devise suitable technical conditions that will reflect the specific environmental risks at each application site.

As outlined in Section 11.3, the environmental risks of Phoslock™ should be viewed as site-specific and this requires the adoption of new application procedures, including direct toxicity assessments of the chemical and determination of pertinent water quality parameters for each site. The use of this approach requires the development of a new pre-application physical, chemical and biological testing regime for Phoslock™ that is dictated by the environmental chemistry of both the chemical and ionic lanthanum that may be released from the product. It will also require the development of more focussed post-application monitoring protocols for the water and sediment compartments of the environmental waters treated. The recommendations for enhanced environmental risk management are discussed in the subsequent section.

11.3 Options for improved risk management

11.3.1 Safety data sheets and labels

Under the *Model Work Health and Safety Regulations* (Safe Work Australia, 2011c) and the Commonwealth, state and territory regulations introduced in accordance with these national model regulations, employees are required to have ready access to Safety Data Sheets for hazardous substances at their workplace since it provides information to those who use the hazardous substance.

Importers of Phoslock™ should review their safety data sheets for compliance with the *Code of Practice for the Preparation of Safety Data Sheets for Hazardous Chemicals* (Safe Work

Australia, 2011b). The safety data sheets should be provided to the occupational health and safety officer during the workplace assessment process.

In accordance with the *Code of Practice for the Labelling of Workplace Hazardous Chemicals* (Safe Work Australia, 2011a) it is recommended that importers of Phoslock™ review their labels for compliance.

It is noted that the current label states that the material is "Non Poisonous – no recommended exposure standards for this product". There is currently an atmospheric exposure standard in existence for non-hazardous 'dusts not otherwise classified' (10 mg/m^3 TWA). The current application methods of Phoslock™ do give rise to a risk of dust generation in the vicinity of workers' breathing space and therefore the statement on the label is considered inappropriate. Exposure standards are not required on labels, but where they are mentioned, the correct information should be provided.

11.3.2 Approvals and conditions for the use of Phoslock™

Every Australian state and territory stipulates controls on releases of contaminants to the environment through legislation, policies and guidance. Table 11.1 lists primary legislations pertaining to environment protection and pollution management.

Table 11.1. Primary environment protection legislation for pollution management.

State/Territory	Relevant Legislation	Principal Authority
Australian Capital Territory	Environment Protection Act 1997 and regulations	Environment and Sustainable Development Directorate
New South Wales	Protection of the Environment Operations Act 1997 and regulations	NSW EPA, and Office of Environment and Heritage
Queensland	Environment Protection Act (1994) and regulations	Department of Environment and Heritage Protection
Northern Territory	Waste Management and Pollution Control Act 1998 and regulations and Water Acts	Department of Natural Resources, Environment, The Arts and Sport
Western Australia	Environmental Protection Act (1986) and regulations	WA Department of Environment and Conservation
South Australia	Environment Protection Act (1993) and regulations	SA EPA
Victoria	Environment Protection Act (1970)	EPA Victoria
Tasmania	Environmental Management and Pollution Control Act (1994)	EPA Tasmania

The following guide to site-specific testing of Phoslock™ should be used as required under the various state and territory legislative and regulatory frameworks.

11.3.3 Site-specific Phoslock™ application matrix

The site-specific direct toxicity assessment procedure is necessary for Phoslock™ given the current uncertainties regarding the environmental fate and toxicity of this chemical and its toxic ionic lanthanum component in aquatic ecosystems. There is a need for pre-application direct toxicity assessments of this chemical provided that certain preliminary physical and chemical water quality parameters that indicate that Phoslock™ use in the water body may be appropriate are met. The minimal release of dissolved lanthanum in environmental waters is highly dependent on physical and chemical water quality parameters. The optimal conditions for Phoslock™ application in soft waters are in the pH range 6 to 8.35, and FRP concentrations >0.1 mg/L. Phoslock™ can be applied in hard waters regardless of the FRP level, before or after an algal bloom crash where the FRP level is maximised.

In the development of direct toxicity assessment (DTA) procedures in Australia, the Water Quality Guidelines (ANZECC/ARMCANZ, 2000a) presented some factors that must be considered which include the following main features: test species selection, dilution water selection, nature of the contaminant, test methodology, biological endpoints, and quality assurance/quality control. The description of each factor is outlined in the Water Quality Guidelines and is proposed for use in determining the site-specific application of Phoslock™.

A matrix has been developed that identifies potential higher or lower risk circumstances in applying Phoslock™. The matrix is presented as a guide since a greater understanding of the interaction of Phoslock™ to environmental waters and aquatic organisms is needed to be able to formulate more specific conditions for the safe use of Phoslock™.

Application conditions with higher risk potential

From the analysis of the water chemistry parameters and limitations in the application of the chemical to environmental waters (Sections 10.3.2 and 10.3.3), the circumstances for which the application of Phoslock™ that may constitute a higher risk include, but are not limited to, the following:

- Softwater/low alkalinity (<60 mg/L $CaCO_3$) and FRP < 0.1 mg/L;
- Softwater/low alkalinity (<60 mg/L $CaCO_3$) and FRP ≥0.1 mg/L.

For water bodies meeting these high risk potential criteria, the presumption should be that Phoslock™ is not used to mitigate FRP, and if Phoslock™ is proposed for use in these circumstances, a detailed justification should be provided. The following DTA requirements should be considered in cases where the detailed justification indicates that use of Phoslock™ could be considered.

Before the application of Phoslock™ in an environmental water body classified as softwater and/or low alkalinity with a FRP concentration less than 0.1 mg/L, further detailed fundamental investigations of the environmental fate and effects of Phoslock™ and ionic lanthanum in the specific water and sediment compartments are needed. In all stages of the pre- and post-application monitoring, the key water chemistry parameters (pH, hardness/alkalinity, total phosphorus and FRP, total and dissolved lanthanum, DO/DOC, chlorophyll-a) are required. The DTA requirements are presented in Table 11.2.

In addition, determining the porewater lanthanum concentrations, and porewater lanthanum fluxes (if any) to overlying waters, and establishing criteria to determine to what extent these pose a risk to ecosystem health are recommended. The porewater concentration of ionic lanthanum is expected to be most closely correlated with toxic effects on detritus-feeding benthic invertebrates and is therefore a high priority for monitoring studies associated with the application of Phoslock™. Consideration will need to be given to how best to determine porewater concentrations of lanthanum as centrifuging the sediment layer will underestimate the porewater concentration. A sediment peeper or DGT (diffusive gradients in thin films) sampler might provide superior results, and the trial of these techniques for measurement of dissolved lanthanum in sediments is desired.

Table 11.2. Site-specific DTA for softwater with FRP level <0.1 mg/L.

DTA Factors	Testing prerequisite for softwater with low FRP
Test species selection	Aquatic species selected should be endemic to the water body being tested but species with economic relevance (e.g. recreational purposes) could also be chosen, with consideration of the sensitive life stages. Testing on at least three taxonomic levels: 1 aquatic plant or algae, 2 invertebrates (1 daphnid and 1 sediment-dwelling species), and 1 fish.
Nature of the contaminant	Systematic investigation of the influence of various water quality properties on the speciation and rates of release of ionic lanthanum from Phoslock™ is needed, including the partitioning of lanthanum in the water column and sediment compartment as a function of time using mass balance measurements of lanthanum.
Test methodology	Pre-application direct toxicity testing of Phoslock™ in local dilution waters and pre- and post-application biological and chemical monitoring of the water body. A parallel testing using ionic lanthanum salts in local dilution waters at comparable concentrations to lanthanum that could occur in the water column if all of the intercalated lanthanum in the applied Phoslock™ granules was released and measuring the levels of dissolved and total lanthanum and key water chemistry parameters.
Test/biological endpoints	Growth endpoint test for algae or plant species; acute and/or chronic immobilisation test for daphnid; chronic reproduction or growth rate test for sediment-dwelling invertebrate; and sub-acute lethality for fish.

The results of these additional studies for softwater with low FRP could be used to derive locally specific trigger values for lanthanum toxicity in the water and sediment compartments. These values will be the maximum permissible dissolved lanthanum concentrations in the water and sediment compartments following application of Phoslock™.

For an environmental water body classified as softwater and/or low alkalinity with a FRP concentration ≥ 0.1 mg/L, the recommended dosing ratio of 100:1 Phoslock™ to FRP should be used to ensure that the ionic lanthanum levels are equivalent to the available phosphorus. The key water chemistry parameters (pH, hardness/alkalinity, total phosphorus and FRP, total and dissolved lanthanum, DO/DOC, chlorophyll-a) are to be measured in all stages of the pre- and post-application monitoring. The DTA requirements are presented in Table 11.3.

The toxicity testing is required based on the Phoslock™ effects observed in tests involving softwater wherein the sensitive species were found to be aquatic invertebrates and fish. Direct toxicity assessments using local dilution waters on the sensitive species are necessary to confirm that Phoslock™ and the ionic lanthanum it liberates will not have adverse effects on biota in the water and sediment compartments of water bodies.

Table 11.3. Site-specific DTA for softwater with FRP level >0.1 mg/L.

DTA Factors	Testing prerequisite for softwater with high FRP
Test species selection	Testing on aquatic invertebrates (1 daphnid and 1 sediment-dwelling species, preferably amphipod) and 1 fish species (locally important fish species or its representative) that is found locally in the water body.
Nature of the contaminant	The highest FRP measured in the water samples should be used to estimate the amount of Phoslock™ that should be applied to the water body based on the recommended dosing of 100:1 (Phoslock™ to FRP).
Test methodology	Pre-application direct toxicity testing of Phoslock™ in local dilution waters and pre- and post-application biological and chemical monitoring of the water body. A parallel testing using ionic lanthanum salts in local dilution waters at comparable concentrations to lanthanum that could occur in the water column if all of the intercalated lanthanum in the applied Phoslock™ granules was released and measuring the levels of dissolved and total lanthanum and key water chemistry parameters.
Test/biological endpoints	Acute and/or chronic immobilisation test for daphnid; chronic reproduction or growth rate test for sediment-dwelling invertebrate; and sub-acute lethality for fish.

Application conditions with lower risk potential

From the analysis of the water chemistry parameters and limitations in the application of the chemical to environmental waters (Sections 10.3.2 and 10.3.3), the circumstances for which the application of Phoslock™ that may constitute a lower risk include, but are not limited to, the following condition:

- Hardwater/high alkalinity (>60 mg/L $CaCO_3$) regardless of FRP level.

For environmental waters classified as hardwater, Phoslock™ has reduced toxicity to aquatic organisms except for some invertebrates (midge and/or amphipod). The FRP concentration is not a limiting factor for hardwaters since any excess ionic lanthanum not captured by the FRP would bind with the carbonate from highly alkaline waters. The key water chemistry parameters (pH, hardness/alkalinity, total phosphorus and FRP, total and dissolved lanthanum, DO/DOC, chlorophyll-a) are to be measured in all stages of the pre- and post-application monitoring, with DTA requirements presented in Table 11.4.

Table 11.4. Site-specific DTA for hardwater regardless of FRP level.

DTA Factors	Testing prerequisite for hardwater with any FRP
Test species selection	Testing on aquatic invertebrates (1 daphnid and 1 sediment-dwelling species, preferably amphipod) endemic to the water body.
Nature of the contaminant	Recommended dosing ratio of 100:1 Phoslock™ to FRP.
Test methodology	Pre-application direct toxicity testing of Phoslock™ in local dilution waters and pre- and post-application biological and chemical monitoring of the water body. A parallel testing using ionic lanthanum salts in local dilution waters at comparable concentrations to lanthanum that could occur in the water column if all of the intercalated lanthanum in the applied Phoslock™ granules was released and measuring the levels of dissolved and total lanthanum and key water chemistry parameters.
Test/biological endpoints	Acute and/or chronic immobilisation test for daphnid; and chronic reproduction or growth rate test for sediment-dwelling invertebrate.

Glossary

NICNAS uses the IPCS Risk Assessment Terminology (IPCS, 2004) glossary which includes Part 1: IPCS/OECD Key Generic Terms used in the Chemical Hazard/Risk Assessment and Part 2: IPCS Glossary of Key Exposure Assessment Terminology. The IPCS Risk Assessment Terminology can be accessed at:

http://www.who.int/ipcs/methods/harmonization/areas/ipcsterminologyparts1and2.pdf

Appendix

The assessment of toxicological data of Phoslock™ as a new chemical (NA/899) is reproduced here (NICNAS, 2001) without modification.

A.1 EVALUATION OF HUMAN HEALTH EFFECTS

There is no toxicological data for the notified chemical. The notifiers provided copies of published scientific literature on the constituent chemicals of lanthanum chloride and bentonite, as surrogate data. Mammalian toxicological data on lanthanum chloride and bentonite were supplied in form of an extensive literature review.

A.1.1 Lanthanum chloride

Lanthanum chloride (CAS No. 10099-58-8), as one of the rare earth chloride salts, may have anticoagulative effects (Beaser, 1942; Dycheroff & Gruenewald, 1943). Lanthanum competes with calcium in a large range of biomolecules and biomolecular processes. A review by Das et al., (1988) reported that La^{3+} reacts *in vitro* with various tissue components, eg proteins, enzymes and phosphates. By displacing and replacing calcium ions in certain selected cell systems, La^{3+} inhibits the significant role of calcium in the various cellular processes. For example, La^{3+} inhibits the calcium pump of red blood cells and, in animal studies, La^{3+} has been shown to inhibit muscle activity by blocking calcium-activated enzymes.

The acute oral toxicity of lanthanum chloride in rats (LD_{50} 2 370-4 184 mg/kg) is very low (Cochran, 1950, Sax, 1984, RTECS, 2000). When giving subcutaneous injections, the LD_{50} was determined to be >1 000, 3 500 and >500 mg/kg in frogs, mice and rats, respectively (Sax, 1984). Information from toxicity studies has indicated that the liver is the target organ (Das et al., 1988).

Lanthanum chloride was non-mutagenic in a bacterial mutagenicity assay (EPA GeneTox Program, 1988). However, intraperitoneal injection of lanthanum chloride caused an increase in the mitotic index and the nuclear volume of liver cells, and an immediate decrease in the mitotic index of rat and mouse bone marrow cells (De and Sharma, 1981; Das et al., 1983). In the review by Das et al. (1983) chromosomal changes have been observed in a number of studies. Dose-related binding to DNA has also been observed.

In the reproductive and developmental toxicity studies, lanthanum chloride caused sperm morphological changes, and reduction of sperm motility and sperm count in goats (RTECS, 2000). A single injection of 44 mg La/kg into pregnant mice reduced the number of successful pregnancies and average litter size (Abramczuk, 1985).

A few studies indicate that rare earth compounds contribute to the risk of pneumoconiosis and chronic pulmonary reactions in workers. However, no direct evidence with lanthanum chloride has been found in workers. In a comparative *in vitro* toxicity study in a pulmonary alveolar macrophage culture, lanthanum chloride gave an LC_{50} of 52 μM, comparable to cadmium (28 μM), suggesting that exposure to lanthanum may be harmful.

Lanthanum and lanthanum salts are not on the NOHSC *List of Designated Hazardous Substances* (NOHSC, 1999a). Based on the available information, they are unlikely to be classified as hazardous substances in accordance with the NOHSC *Approved Criteria for Classifying Hazardous Substances* (NOHSC, 1999).

A.1.2 Bentonite

Bentonite (CAS No. 1302-78-9) consists of a group of clays formed by crystallisation of vitreous volcanic ashes that were deposited in water. It has been used as a filler in crayons, a

lubricant in oil well drilling, a base in cosmetics and in the manufacture of concrete. Bentonite has been approved as a food additive in Australia.

The expected acute oral toxicity of bentonite in humans is very low (LD_{50}>15 g/kg) (HSDB, 2000). However, severe anterior segment inflammation, uveitis and retrocorneal abscess from eye exposure were reported when bentonite had been used as a prophypaste (Austin & Doughman, 1980).

In a 33 day dietary (2 and 6%) and a 90 day dietary (1, 3 and 5%) studies in chickens, no changes in behaviour, overall state, clinical and biochemical parameters and electrolytic composition of the blood. Repeat dietary administration of bentonite did not affect calcium or phosphorus metabolism. However, larger amounts caused decreased growth, muscle weakness, and death with marked changes in both calcium and phosphorus metabolism.

Bentonite did not cause fibrosis after 1 year exposure of 60 mg dust (<5 μm) in a rat study (Tatrai, 1985). However, in a second rat study, where 5 μm particles were intratracheally instilled at 5, 15 and 45 mg/rat, dose-related fibrosis was observed. Bentonite clay dust is believed to be responsible for bronchial asthma in workers at a processing plant in USA (Browning, 1969).

Ingestion of bentonite without adequate liquids may result in intestinal obstruction in humans. Hypokalemia and microcytic iron-deficiency anaemia may occur in patients after repeat doses of clay. Chronic ingestion has been reported to cause myositis (HSDB, 2000).

Bentonite is not on the NOHSC *List of Designated Hazardous Substances* (NOHSC, 1999a). Based on the available information, it is unlikely to be classified as a hazardous substance in accordance with the NOHSC *Approved Criteria for Classifying Hazardous Substances* (NOHSC, 1999).

A.1.3 Lanthanum modified clay (Phoslock™)

No toxicological data were provided for the notified chemical.

The notified chemical is a bentonite clay partially substituted with lanthanum. CSIRO unpublished data indicate that Phoslock™ closely resembles bentonite in its physical properties with a single exception; Phoslock™ does not appreciably swell on absorption of water. In addition, free lanthanum ions are not dissociated from Phoslock™ when the latter is placed in an aqueous environment. Phoslock™ is a colloidal hydrated aluminium silicate which will form highly viscous suspensions or gels in water. These colloidal particles are unlikely to across the biological barriers.

Phoslock™ is expected to have a LD_{50} >15 g/kg in humans with adequate water intake. The MSDS for Phoslock™ states that eye contact may cause mild transient physical irritation due to the presence of clay particles. Phoslock™ is not expected to be a skin irritant.

Inhalation exposure to dusts from dried bentonite clay or dried Phoslock™ may cause chronic lung disease such as bronchial asthma and there is information to suggest that the rare earths may contribute to the risk of pneumoconiosis and chronic pulmonary reactions in workers. In addition, as Phoslock™ may be manufactured from bentonite containing contain crystalline silica, silicosis may result after chronic inhalation. Therefore, inhalation of lanthanum modified bentonite dust may lead to respiratory illness.

Due to the proportions of lanthanum and bentonite in the notified chemical, the toxicity profile is likely to resemble that of bentonite. Based on the available information and the low toxicity of bentonite, Phoslock™ is unlikely to be classified as a hazardous substance according to NOHSC *Approved Criteria for Classifying Hazardous Substances* (NOHSC, 1999).

A.2. ASSESSMENT OF ENVIRONMENTAL EFFECTS

Ecotoxicological data were supplied in the form of an extensive array of tests, using predominantly USEPA TCLP (Toxicity Characteristic Leach Protocol) evaluation scheme.

The initial sets of data were laboratory studies on fish, daphnia and algae conducted with leachate solutions prepared from the clay (Section A.2.2.).

After application of the clay to an 800 metre stretch of the Canning river (WA) in two large scale field trials conducted in January and April 2000, a large number of toxicity tests (fish, *Ceriodaphnia* and algae) were conducted with the river water collected before and after application (Section A.2.3.). In addition to these toxicity data, in situ observations of resident biota were also conducted before and after clay application.

It is important to note that the trial conducted in April 2000 utilised lanthanum exchanged clay from which most of the readily leachable free lanthanum had been removed through a prior chemical conditioning process during manufacture. It was reasoned that reduced lanthanum concentration in the water column after clay application would reduce the potential for toxic effects arising from this element, and the results of post-application monitoring appeared to confirm this.

In conjunction with these tests, a series of lanthanum calibration bio-assays for fish, *Ceriodaphnia* and algae were also conducted in order to establish realistic toxicities of lanthanum against these species (Section A.2.4.).

These studies are discussed in detail below.

A.2.1. Initial Laboratory Tests with Leachate Prepared from Modified Clay

The initial laboratory data were generated using leachate solutions prepared in synthetic soft water and/or de-ionised water (Milli-Q, reverse osmosis) derived from the modified clay and are summarised in the following table. The test procedures and results obtained are described and the implications of the observed toxicity to some species are discussed. The tests were performed in compliance with OECD/EEC Test Methods and according to OECD Principles of Good Laboratory Practices.

The leachate solutions were prepared using the TCLP (Toxic Characteristic Leachate Procedure) method developed by the US EPA. In this method the solid test material (50 grams of a laboratory preparation of the modified clay) was tumbled in a teflon bottle for 18 hours with 1 L of either the synthetic soft water (48 mg/L $NaHCO_3$, 30 mg/L $CaSO_4.2H_2O$, 30 mg/L $MgSO_4$ and 2 mg/L KCl; hardness equivalent to 40-48 mg/L $CaCO_3$; alkalinity equivalent to 30-35 mg/L $CaCO_3$) or de-ionised (reverse osmosis) water. Following the tumbling procedure the liquor was filtered through a 45 µm filter and used in the toxicity tests.

Laboratory Ecotoxicity Test Results

Test	Species	Water type	Results
Sub-acute Toxicity (Imbalance Static Test) (OECD TG 203)	Eastern rainbow fish *Melanotaenia duboulayi*	Synthetic soft water	96 h EC_{50} >100% leachate (127 μg/L La – see notes below).
Acute Toxicity -Immobilisation (Static Test) (OECD TG 202)	Water Flea (*Ceriodaphnia dubia*)	Synthetic soft water	48 h EC_{50} = 49% leachate (80 μg/L of La.) NOEL = 25% leachate.
Acute Toxicity -Immobilisation (Static Test) (OECD TG 202)	Water Flea (*Ceriodaphnia dubia*)	Milli-Q water	48 h EC_{50} = 10% leachate (approximately 40 μg/L of La.)
Chronic Toxicity – (Static Test) (OECD TG 202)	Water Flea (*Ceriodaphnia dubia*)	Synthetic soft water	7 day EC_{50} = 41% leachate (12 μg/L of La) NOEC <6.25% leachate
Growth Inhibition (OECD TG 201)	Green Algae (*Selenastrum capricornutum*)	Milli-Q water	Promoted algal growth at all leachate concentrations ≥ 6.25% – see notes below.

Fish (Lim, 1999; Stauber, 2000)

The sub-acute toxicity test on fish was performed over a 96 hour period in leachate solutions prepared from synthetic soft water (hardness 40-45 mg/L as $CaCO_3$) using a static methodology without replacement of the test media. The test was conducted at 22.4±1.3°C using test concentrations containing 0 (control), 12.5, 25, 50, 75 and 100% of the leachate made up by mixing the appropriate volume of the 100% leachate with synthetic soft water. Four juvenile eastern rainbow fish were used in each test, and their general behaviour and appearance was monitored over the 96 hour test period. Throughout the test the dissolved oxygen was always between 79 and 108% saturation, while pH was always between 7.15 and 7.7.

In the 75% leachate and lower concentrations, no imbalance in swimming was observed over the 96 hour test period, but one fish (out of the four) in the 100% leachate exhibited some imbalance after 96 hours.

The conclusion from these results is that the soft water leachate from the modified clay has a Lowest Observed Effect Level (LOEL) of 100% leachate for this fish species although the EC_{50} >100%. The leachate was analysed for Zn, Cu and La and found to contain 127 μg/L, <1 μg/L and 127 μg/L of each of these elements, respectively.

Aquatic Invertebrates – Cladoceran (Stauber, 2000)

Acute toxicity tests were conducted over 48 hour periods against *Ceriodaphnia dubia* at 25±1°C using leachate solutions prepared with synthetic soft water, ultrafiltered synthetic soft water and Milli-Q water. The test concentrations were performed at 0 (control), 6.25, 12.5, 25, 50, 75 and 100% of leachate concentrations, with four replicate tests carried out at each concentration using five test animals in each test vessel (ie. 20 daphnia exposed to each test concentration).

The leachate in synthetic soft water exhibited significant toxicity to this species, with the 48 hour EC_{50} calculated from the observed immobilisation data as 49% leachate, with the corresponding No Observed Effect Concentration (NOEC) being 25% leachate. The soft water leachate used in this test was analysed and found to contain 163 μg/L lanthanum, and 49% leachate would correspond to approximately 80 μg/L of lanthanum.

Toxicity tests on *Ceriodaphnia dubia* using a leachate prepared using Milli-Q (reverse osmosis) water and the solutions were found to be more toxic than for the soft water leachates, with a 48 hour LC_{50} of 10% leachate (compared with the Milli-Q water control). The lanthanum concentration was 396 µg/L, so 10% leachate would correspond to around 40 µg/L of lanthanum. However, interpretation of the results was not straight forward because there was significant mortality in the control water (ie. Milli-Q water) which suggested that threshold concentrations of some ions (possibly Ca^{2+}) in the water are required for normal survival of this species. The report suggested that this species of cladoceran (*Ceriodaphnia dubia*) will not survive in water with conductivity lower than 100 µS/cm.

A chronic (reproduction) test against *Ceriodaphnia dubia* was also conducted over a seven day period using diluted leachate from a separate batch of synthetic soft water leachate prepared from a different laboratory preparation of the clay. Seven day survival of the daphnids (10 animals for each concentration) was significantly reduced for all tested leachate concentrations above 25%, and the 7 day EC50 was calculated as 41% leachate. There was also a significant reduction in the number of young daphnids produced at all tested leachate concentrations, giving a NOEC <6.25% leachate. This particular soft water leachate was analysed for lanthanum and found to contain 2.01 mg/L after filtration, so the 41% EC_{50} corresponds to approximately 820 µg/L of lanthanum. When compared with the acute 48 h EC_{50} of approximately 80 µg/L, this result is surprising, and is discussed below.

Algae (Stauber, 2000)

A test on the toxicity of Milli-Q water leachates on the growth of green algal (*Selenastrum capricornutum*) biomass at 24±2°C was conducted over a three day period. Surprisingly rather than showing toxic effects, the algae grew significantly faster in the leachate than in the Milli-Q water controls. The test was conducted using 0 (control), 6.25, 12.5, 25, 50, 75 and 100% leachate, and the test at each concentration was conducted in triplicate.

For all the leachate test solutions, the growth rate of the algae was approximately twice that of the control suggesting that some component of the leachate acts as a growth promoter for this species.

Although the lanthanum concentration was not reported for this leachate, it is assumed to be the same as for the Ceriodaphnia dubia test in Milli-Q water, so 6.25% leachate would correspond to a lanthanum concentration of around 25 µg/L.

Discussion

It is reasonable to assume that any toxic properties of the leachate solutions are most likely associated with leached lanthanum. The toxic properties of the leachate solutions derived from the new lanthanum modified clay are unusual in that –

a) fish are apparently not affected, or only slightly affected (lanthanum concentration around 127 µg/L),

b) *Ceriodaphnia dubia* are extremely sensitive to acute exposure, and the 48 hour EC_{50} in synthetic soft water was determined as 49% of the leachate concentration which corresponded to a lanthanum concentration of around 80 µg/L. However, in a chronic (seven day) survival and reproduction test, the EC_{50} was determined as 41% of leachate concentration which in this case corresponded to a lanthanum concentration of around 820 µg/L. These results appear contradictory since it is usual for chronic end points to be lower than acute ones. In the present case it must be assumed that differences in the composition (eg hardness, HCO^{3-} concentration) of the water used in preparing the leachates in the two test types was responsible for the seven day reproduction EC_{50} being significantly higher than the acute 48 hour EC_{50}. Unfortunately no comprehensive analyses of water chemistry were provided to substantiate this possibility.

c) Green algal growth is apparently promoted in the leachate, and the test results suggest that lanthanum concentrations as low as 6.25% of the leachate concentration (lanthanum approximately 7 µg/L) stimulate growth of *Selenastrum capricornutum*.

The *Ceriodaphnia dubia* results are in general accord with those of an independent Australian study on the acute and chronic toxicity of lanthanum chloride to *Daphnia carinata* (Barry and Meehan, 2000), with these workers finding an acute 48 hour LC_{50} of 43 µg/L in soft water (hardness equivalent to 22 mg/L as $CaCO_3$). However, the toxicity was reduced in harder water and the LC_{50} in ASTM water (hardness = 160 mg/L as $CaCO_3$) was 1180 µg/L. However, Barry and Meehan (2000) reported chronic effects (7 day) at concentrations >40 µg/L in both the soft water and ASTM water. It is important to note that the tests of Barry and Meehan (2000) did not occur in the presence of humic substances which are likely to ameliorate the effects of free lanthanum in solution.

In respect of promotion of algal growth, it is of interest that agricultural applications of rare earth salts in China have been reported to significantly increase crop production (eg. Tribe et al., 1990).

It should also be pointed out that the amount of free lanthanum released from the clay in these leachate tests is only a very small fraction of the total lanthanum contained in the clay. For example, the Milli-Q water leachate contained a concentration of lanthanum which is estimated to correspond to a loss of less than 0.02% of the total lanthanum contained in the clay.

The notifiers provided a hazard quotient (Q = estimated environmental concentration/effect concentration of most sensitive species) of <0.004. However, assuming that the clay is applied at a typical application rate to one hectare of water of depth 1 metre and assuming the highest leaching rates encountered in the laboratory tests, then the derived PEC in the water is estimated as around 20 µg/L.

Using the *Ceriodaphnia dubia* EC_{50} of 80 µg/L, this provides a hazard quotient of 0.25. This scenario is considered to be a very much worst case one, and as indicated previously improved production techniques are expected to eliminate or significantly reduce the amount of free lanthanum leached from the clay and this will lead to a concomitant increase in the safety factor. In addition it is likely that the free ion concentration of La in a waterbody will be further reduced due to precipitation with dissolved phosphate, adsorption to particles and/or complexation with humic substances, thus further decreasing the hazard quotient.

A.2.2. Tests in "treated" Canning River Water

Fish (Stauber and Binet, 2000)

The sub-acute toxicity test on fish was performed over a 96 hour period in 76 samples (40 from Phoslock™ treated sites, and 36 from control sites) of the Canning River water taken from a number of sites after the second field application of the clay in April 2000. Samples were collected prior to clay application, on the day of application and on days 1, 3 and 7 after application. Four replicate tests were conducted for each sample, using 5 juvenile eastern rainbow fish for each test (ie. 20 fish used in each test). No renewal of test media was performed, although gentle air sparging ensured that the dissolved oxygen levels were always >60% of O_2 saturation.

No significant imbalance of the fish was observed in most of the test samples over the 96 hour test period, although up to 15% of the fish (ie. 3 of 20) showed some effects after 96 hours. These data indicate that the clay treatment of the river did not contaminate the water with residues at levels sufficient to produce acute toxic effects in this species. Unfortunately, although pH, dissolved oxygen and temperature were monitored throughout the tests, the level of dissolved lanthanum in the samples was not reported.

In a supplementary lanthanum bio-assay test conducted with solutions of lanthanum chloride made up in water at lanthanum concentrations between (nominally) 750 µg/L and 48 µg/L, 100% mortality of eastern rainbow fish was found for all nominal lanthanum concentrations, indicating a 96 hour LC50 significantly less than the nominal 750 µg/L (measured as 600 µg/L) (see Section A.2.4.1).

Consequently, all that can be concluded from these results is that the samples collected from the Canning River sites treated with the new clay contained available lanthanum at significantly less than the nominal 750 µg/L (600 µg/L measured). The report mentioned that it was not possible to perform tests at lower concentrations due to the shortage of rainbow fish.

Aquatic Invertebrates – Cladoceran (Stauber and Binet, 2000)

Acute tests

Samples of the river water were taken from four sites before and after (on the first day, and then 1, 2, 3 and 4 weeks after application) the first application of the modified clay (January 2000), and these samples were used in laboratory toxicity (immobilisation) tests conducted against *Ceriodaphnia*. Some acute toxicity was observed in all samples collected on the first day after application (up to 29% compared with controls), although the water samples taken one week and later after clay application did not exhibit toxicity.

Although a detailed breakdown of lanthanum concentrations in the samples was not provided in the report, for the first day samples which exhibited some toxicity, a rough correlation was apparently established between the lanthanum concentration and degree of immobilisation. It was remarked that – with two exceptions – only those water samples with total lanthanum concentrations in excess of the 2.6 mg/L NOEC established in the bio-assay calibration tests (see further below) were toxic.

The tests were repeated after the second application of clay in April 2000 with water samples taken on the day of application and 1, 3 and/or 7 days after application. This trial was conducted with lanthanum exchanged clay which had been treated to remove most of the available "free" metal. No toxic effects were observed in any of these samples, but it was remarked that the lanthanum concentrations in the water never exceeded 1.7 mg/L which was less than the above NOEC of 2.6 mg/L.

Chronic tests

In the first field trial (January 2000), chronic toxicity tests of the river water (post clay application) to *Ceriodaphnia* were apparently conducted only with the samples collected the first day after application. The tests were conducted over 7 days and all water samples were toxic, giving reduced survival and reproduction rates compared with the controls. In some cases the number of young produced was <1 per female.

In the second trial (April, 2000) river water was collected on the day of application and 1, 3 and/or 7 days after application. No chronic toxicity to *Ceriodaphnia* was observed. The mean number of young produced per female was between 81 and 149% of the controls, despite the lanthanum concentration in some of the water samples exceeding the 0.09 mg/L LOEC established in the bio-assay calibration tests (Section A.2.4.).

Algae (Stauber and Binet, 2000)

Samples of the river water were taken from a number of sites on the first day, and 1, 2, 3 and 4 weeks after the first clay application in January 2000, and used in laboratory algal growth tests. Although the lanthanum concentrations in the samples were not provided in the report these were apparently measured and on the first day after application lanthanum concentrations as high as 3 mg/L were recorded in water from some of the sampling sites. However, mostly the water showed no inhibitory effect on algal growth even when lanthanum concentrations exceeded the LOEC of 0.13 mg/L (Section A.2.4.). In only one case (water taken from site designated P3) was significant toxicity observed in bottom and

surface water samples taken on the first day after application and measured lanthanum concentrations were very high at 11-15 mg/L. However, it was mentioned in the report that the growth inhibition may have been caused by the removal of bio-available phosphorus rather than reflecting the true toxicity of the lanthanum. This possibility is discussed in Section 10.3.3.

A similar series of tests was conducted after the second application of clay in April 2000, with water samples being taken on the day of application then 1, 3 and/or 7 days after application. Again, no toxic effects were observed, but stimulation of algal growth was between 65 and 450% of controls.

Discussion

Most tests showed that toxic effects to *Ceriodaphnia* were observed in water samples taken on the first day after the first application of the modified clay to the river.

Lesser toxic effects were observed for green algae, but in all cases the toxicity was attributed to free lanthanum liberated from the modified clay. In water samples taken one week and more after clay application, toxic effects were absent.

In the second application the clay had been treated to mitigate release of residual free lanthanum, and the available information indicates that this treatment did reduce release of lanthanum, although concentrations as high as 1.7 mg/L were measured in some samples.

Nevertheless, no toxic effects were observed in fish, *Ceriodaphnia* or algae, despite the measured total lanthanum concentrations sometimes exceeding the NOEC for the test species. This is attributed to mitigation of lanthanum toxicity through the prior removal of residual free (leachable) lanthanum during the manufacturing process.

A.2.3. Bio-Assay Calibration Tests

These tests were conducted in the laboratory in order to establish toxic levels of lanthanum in synthetic soft water (prepared as indicated in the previously described leachate tests) to representative fresh water species of fish, invertebrates and green algae.

Laboratory Bio-assay Results

Test/test media	Species	End Point	EC_{50}** (mg/L)	LOEC (mg/L)	NOEC (mg/L)
Fish/not specified	*Melanotaenia duboulayi*	96 h (Immobilisation*)	<0.6	<0.6	<0.6
Invertebrates/ artificial soft water	*Ceriodaphnia dubia*	48 h (Immobilisation)	5		2.6
		7 day (Immobilisation)	0.51		
		7 day (Reproduction)	0.43	0.09	0.05
Algae/algal test media	*Selenastrum capricornutum*	72 h (Growth inhibition)	0.45	0.13	<0.13

*It was not clear from the report whether this end point was immobilisation (imbalance) or true fish mortality.

**In all tests apart from those on fish these end point lanthanum concentrations refer to total La, which includes both truly dissolved lanthanum and particulate lanthanum. Analytical data tabulated in the report strongly suggested that much of the lanthanum was present in the test media as fine particles.

Fish

In the tests on rainbow fish (Lim, 2000), it appears that solutions of lanthanum chloride were prepared containing lanthanum at nominal concentrations between 0.75 and 48 mg/L. Five juvenile Eastern rainbow fish were placed in vessels containing 500 mL of each solution and the behaviour noted over a 96 hour period. Test media were not renewed, and over the 96 hour test period the temperature, pH and dissolved oxygen levels were

23.4-24.5°C, 6.5-8.1 and 85-106% saturation, respectively while water conductivity was reported between 997 and 1064 S/cm.

No detailed description of test results was provided, but it appears that all fish were immobilised (or dead) at all test concentrations after the 96 hour test period, indicating that the 96 hour LC_{50} of lanthanum for this species was less than the (nominal) 0.75 mg/L. The report did not specify whether the water used to prepare the solutions was de-ionised (reverse osmosis) or a synthetic water made up for the test.

No detailed analysis of the water was provided, except the measured lanthanum concentration in the nominally 0.75 mg/L solution was given as 0.6 mg/L.

It is not possible to reach any definite conclusions from these data except that, with an LC_{50} <0.6 mg/L, lanthanum is at least highly toxic to this fish species in this water (Mensink et al., 1995).

Ceriodaphnia dubia

The tests on *Ceriodaphnia dubia* (Stauber and Binet, 2000) were conducted over a 48 hour period in artificial soft water (48 mg/L $NaHCO_3$, 30 mg/L $CaSO_4.2H_2O$, 30 mg/L MgSO4 and 2 mg/L KCl; hardness equivalent to 40-48 mg/L $CaCO_3$; alkalinity equivalent to 30-35 mg/L $CaCO_3$) using six nominal lanthanum concentrations between 0 (control) and 23 mg/L. Four replicates vessels at each test concentration were employed, with each vessel initially containing 5 daphnia. After 48 hours exposure to a (nominally) 2.6 mg/L solution of lanthanum, 28% of the daphnia were immobile, increasing to 44% at (nominally) 7.6 mg/L and 94% at 23 mg/L. Probit analysis provided a nominal EC_{50} of 5 mg/L lanthanum and a corresponding NOEC of 2.6 mg/L. However, precipitation of the lanthanum was apparent and after 48 hours, the measured values of total lanthanum in the solutions were always close to the nominal concentrations, however those in filtered samples (filter pore size not specified) were usually <10 µg/L. This indicates that although most of the lanthanum in the test media was present associated with fine particles, the lanthanum appears to be assimilable to the animals in this form and produces toxic effects. Alternatively, the observed toxicity may be due to physical effects, eg irritation of sensitive animal organs by fine particles of insoluble lanthanum salts.

A chronic reproduction test on this species was conducted over a 7 day period using solutions of lanthanum (measured on day one of the test) between 0 (control) and 0.62 mg/L. The lanthanum solutions were found to significantly inhibit reproduction relative to the controls, with 16.5 young produced from each original female daphnid after 7 days for the control, compared to 11.6 young per individual at 0.34 mg/L and 2.4 young at 0.62 mg/L. The 7 day EC_{50} for reproduction was determined as 0.43 mg/L with the corresponding 7 day NOEC was 0.05 mg/L. These concentrations are based on the measured lanthanum concentrations (total lanthanum) determined on day 1 of the test, and as with the acute data above, it is likely that much of the lanthanum was present in association with fine particles.

Algae

A test for lanthanum inhibition of green algal growth (Stauber and Binet, 2000) was conducted over a 72 hour period, using *Selenastrum capricornutum* and seven nominal lanthanum concentrations between 0 (control) and 8.1 mg/L in algal test medium. The tests will not be described in detail, but the solutions inhibited algal growth with an apparent 72 hour EbC_{50} for lanthanum calculated from the growth data as 0.45 mg/L. However, this result must be treated with caution because, as with the *Ceriodaphnia dubia* tests, precipitation of lanthanum took place over the test period. The algal test medium contained 0.57 mg/L PO_4 (which is required as a nutrient for algal growth), and there is a strong possibility that the observed toxicity may be an indirect effect associated with removal of available PO_4 from the growth medium through precipitation as $LaPO_4$.

Discussion

It is clear from the above that interpretation of toxicity data for lanthanum to aquatic species is difficult, and the results appear to be strongly dependent on the chemical composition of the water or test media used.

In particular, under the pH regimes used in toxicity tests (typically 7-8), lanthanum phosphate and other salts are highly insoluble so interpretation of toxicity data can be confounded through at least two effects –

a) Uncertainty in true lanthanum toxicity due to its removal through association with particulate material, and
b) The possibility of "growth inhibition" caused through removal of nutrients such as PO_4 from the water through interaction with lanthanum.

In the present series of tests it is not possible to make any definite statement in regard to fish toxicity since the quality of the water was not specified. The 96 hour LC_{50} of <0.6 mg/L found in the present tests indicates that lanthanum may be at least highly toxic to fish in some water.

In *Ceriodaphnia dubia*, the acute 48 hour LC_{50} of 5 mg/L and chronic 7 day EC_{50} of 0.43 mg/L indicate toxicity. However, removal of dissolved lanthanum through precipitation was noted and quantified and it is unclear whether the observed toxicity was due to assimilation of dissolved lanthanum or particulate lanthanum. In respect of this, the results of Barry and Meehan (1997) show that the 48 hour LC_{50} of dissolved lanthanum to some daphnia species may be as low as 43 µg/L in soft water.

The green algal test results were confounded by probable removal of PO_4 from the test media by La^{3+}. Although toxic effects were observed, with an apparent 72 hour EbC_{50} for lanthanum of 0.45 mg/L, it is possible that this was an indirect effect caused through the un-availability of phosphorus to the algae.

A.2.4. Conclusions from Acute and Chronic Ecotoxicity Results

There seems little doubt that dissolved lanthanum has at least high acute and chronic toxicity to fresh water fish and to various species of daphnia in soft water, although water quality parameters appear to have a very large effect on the toxicity. In sufficiently hard water free lanthanum may be precipitated reducing lanthanum availability to aquatic species and mitigating toxicity.

Similarly, the lanthanum ion is expected to have high affinity for the negatively charged humic material present in most natural waters. This mechanism will also remove lanthanum from the water column.

In the toxicity data available from Canning River water treated with the modified clay, no toxic effects were observed, although total lanthanum concentrations were occasionally higher than the NOEC values determined in soft water. The data presently available indicates that the amount of free lanthanum likely to be released from the clay after it has been applied to water bodies is very dependent on the chemical procedures used during clay preparation, and improved preparation procedures can give the clay little tendency for lanthanum release.

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