Independent Expert Scientific Committee on Coal Seam Gas and Large Coal Mining Development

Australian Government

Department of the Environment

Knowledge report

Temperate Highland Peat Swamps on Sandstone: ecological characteristics, sensitivities to change, and monitoring and reporting techniques

This report was commissioned by the Department of the Environment on the advice of the Interim Independent Expert Scientific Committee on Coal Seam Gas and Coal Mining and prepared by Jacobs SKM.

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Acknowledgements

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Disclaimer

Contents

Tables

Figures

Summary

Key points

- The Temperate Highland Peat Swamps on Sandstone community (THPSS) is listed as an endangered ecological community under the *Environment Protection and Biodiversity Conservation Act 1999* and is also listed as endangered under the New South Wales *Threatened Species Conservation Act 1995.*
- The THPSS ecological community can be categorised and described using three conceptual types: headwater swamps, valley infill swamps and hanging swamps.
- Bayesian belief network (BBN) modelling was used to model the sensitivity of these swamp types to environmental change. The BBN modelling showed that ecological sensitivity was most strongly influenced by an altered inundation regime.
- A monitoring programme that aims to identify impacts early so that management can be adapted must focus on the subsidence or hydrological impacts since these precede any ecological response.
- Information linking subsidence effects to ecological impacts is limited, with little information that specifically describes how swamp ecology responds to changes in the surrounding environment.
- In light of this, a multiple before–after control–impact (M-BACI) approach to monitoring swamp ecology is proposed as an appropriate basis for designing an ecological monitoring programme.
- A monitoring programme that is capable of detecting impacts to the swamps and attributing the impacts to a specific cause (e.g. longwall coal mining) must incorporate three phases of monitoring, with associated reporting:
  - Phase 1—baseline characterisation
  - Phase 2—assessment of risks
  - Phase 3—ongoing impact monitoring.

The Temperate Highland Peat Swamps on Sandstone (THPSS) ecological community can be categorised and described using three conceptual models:

- headwater swamps—formed near catchment divides where topographic gradients are shallow. These swamps are predominantly reliant on rainfall and run-off
- valley infill swamps—occur in steeper topographies filling the valleys of incised second or third-order streams. These swamps are more likely to be connected to either perched or regional aquifers
- hanging swamps—occur on steep valley sides where there is groundwater seepage.

The fundamental differences between each type are topographical location and the resulting potential for connection to groundwater. The three conceptual models are broad categorisations of the swamps. In reality, a single swamp may be best described by a combination of the conceptual models, particularly as larger swamps can grade from one type to another. Valley infill and hanging swamps are more vulnerable to subsidence impacts, as nonconventional subsidence affects cliffs and steeper topography terrain.

At least 19 threatened species protected under New South Wales and Commonwealth legislation are known to occur within the community, with 14 of these being flora species.

Sensitivity analysis

Bayesian belief networks (BBNs) were used to model the sensitivity of the THPSS to changes in a range of environmental factors as a result of longwall coalmining impacts. The BBNs modelled the sensitivity of the community and individual species. This exercise found that both the community and the individual species were most sensitive to changes in peat stability (such as erosion of the peat), reduced periods of inundation, and an increased frequency and intensity of fire. The potential for changes in peat stability and fire risk are strongly influenced by changes to the inundation regime, which can occur relatively rapidly in response to subsidence. An altered inundation regime therefore has the overall strongest influence on sensitivity for most species in the BBN modelling.

The BBNs developed for this project are based on conceptualisation by specialists rather than on any measurement of impacts and should therefore be used as a risk assessment tool, rather than a definitive measurement of impact. They also provide a framework that can be updated in the future as empirical evidence of impacts to peat swamps becomes available.

Monitoring

A monitoring programme that aims to identify impacts early so that management can be adapted must focus on the subsidence or hydrological impacts since these precede any ecological response. By the time an ecological impact is detected, subsidence effects, hydrological impacts and (potentially) peat destabilisation will have already occurred. It will then be too late to mitigate impacts or to implement adaptive management to minimise impacts on the swamps. A fundamental principle of monitoring ecological impacts of subsidence is therefore to provide an early indication of potential ecological impacts by integrating ecological monitoring with monitoring of subsidence effects and hydrological impacts.

Information linking subsidence effects to ecological impacts is limited, with little information that specifically describes how swamp ecology responds to changes in the surrounding environment. There is also very little understanding of the natural variations in swamp ecology over time. These knowledge gaps mean that current monitoring programmes are not designed to measure specific ecological changes that are known to occur in response to subsidence. Because of this, monitoring is usually unable to distinguish between changes due to natural ecological variation and changes caused by subsidence.

The limited knowledge of swamp variability and ecological responses to subsidence indicates that an appropriate basis for designing an ecological monitoring programme is to adopt a multiple before–after control–impact (M-BACI) approach to monitoring swamp ecology.

A monitoring approach has been developed to maximise the potential for impacts to be observed, and to be accurately attributed to a specific cause (e.g. longwall mining) as opposed to natural variations like drought, seasonal variations or fire. As such, the recommended approach incorporates a significant baseline monitoring programme, beginning at least two years before longwall mining, which aims to monitor the ecological response to subsidence impacts. This cannot be defined in a generic manner; hence, a key requirement of an ecological monitoring programme is to include an extensive baseline monitoring programme that establishes natural variability so that natural variations in ecology can be distinguished from variation caused by subsidence impacts.

A monitoring programme that is capable of detecting impacts to the swamps and attributing the impacts to a specific cause (e.g. longwall coalmining) as opposed to natural variation must incorporate three phases of monitoring, where the outcome of each phase of monitoring informs the design of the subsequent monitoring phase:

- Phase 1—baseline characterisation of swamp ecology
- Phase 2—assessment of risks and acceptable levels of impact
- Phase 3—ongoing impact monitoring programme.

When there has been sufficient baseline monitoring to characterise the ecology of the swamps and to understand the magnitude of natural variability in health and composition (phase 1), an informed risk assessment that helps define the acceptable levels of impact can be done (phase 2). The outcomes of phases 1 and 2 directly control the design of the ongoing monitoring programme for phase 3, in which the parameters to be included should be those that were observed to be responsive to change and for which natural variability was well defined by baseline monitoring.

The duration of ongoing impact monitoring depends on the species present and their impact response times. If hydrological impacts have been observed, the ecological response may continue to progress for many years. Ongoing impact monitoring should occur until both hydrological and ecological monitoring indicate that the system is stable.

Abbreviations

General abbreviations	Description
ANZECC	Australia and New Zealand Environment Conservation Council
ARMCANZ	Agriculture and Resource Management Council of Australia and New Zealand
BACI	Before–after control–impact
BBN	Bayesian belief network
cm	Centimetre
CSIRO	Commonwealth Scientific and Industrial Research Organisation
EPBC Act	*Environment Protection and Biodiversity Conservation Act 1999*
EVI	Enhanced vegetation index
FPC	Foliage projective cover
GPS	Global Positioning System
ha	Hectare
IESC	Independent Expert Scientific Committee on Coal Seam Gas and Large Coal Mining Development
InSAR	Satellite interferometric synthetic aperture radar
km^2	Kilometre squared
L/s	Litre per second
m	Metre
M-BACI	Multiple before–after control–impact
mg/L	Milligram per litre
NDMI	Normalised difference moisture index
NDVI	Normalised difference vegetation index
NDWI	Normalised difference water index
NSW	New South Wales
OWS	Office of Water Science
ppm	Parts per million
SAR	Synthetic aperture radar
THPSS	Temperate Highland Peat Swamps on Sandstone
TSC Act	*Threatened Species Conservation Act 1995* (NSW)
µg/L	Microgram per litre
µS/cm	Micro-Siemens per centimetre

Glossary

Term	Description
Adsorption	The reversible binding of molecules to a particle surface. This process can bind methane and carbon dioxide, for example, to coal particles.
Alkalinity	The quantitative capacity of aqueous media to react with hydroxyl ions. The equivalent sum of the bases that are titratable with strong acid. Alkalinity is a capacity factor that represents the acid-neutralising capacity of an aqueous system.
Anthropogenic	Relating to, or resulting from, the influence of human beings on nature.
Aperture	Separation distance between two fracture surfaces, used as measure of fracture width.
Aquifer	Rock or sediment in formation, group of formations or part of a formation, that is saturated and sufficiently permeable to transmit quantities of water to wells and springs.
Aquitard	A saturated geological unit that is less permeable than an aquifer and incapable of transmitting useful quantities of water. Aquitards often form a confining layer over an artesian aquifer.
Artesian	Pertaining to a confined aquifer in which the groundwater is under positive pressure (i.e. a bore screened into the aquifer will have its water level above ground).
Aquatic ecosystem	Any watery environment from small to large, from pond to ocean, in which plants and animals interact with the chemical and physical features of the environment.
Bore/borehole	A narrow, artificially constructed hole or cavity used to intercept, collect or store water from an aquifer, or to passively observe or collect groundwater information. Also known as a borehole, well or piezometer.
Casing	A tube used as a temporary or permanent lining for a bore. *Surface casing*: the pipe initially inserted into the top of the hole to prevent washouts and the erosion of softer materials during subsequent drilling. Surface casing is usually grouted in and composed of either steel, PVC-U or composite materials. *Production casing*: a continuous string of pipe casings that are inserted into or immediately above the chosen aquifer and back up to the surface through which water and/or gas are extracted/injected.
Compaction	The process by which geological strata under pressure reduce in thickness and porosity, and increase in density.
Compression	A system of forces or stresses that tends to decrease the volume or shorten a substance, or the change of volume produced by such a system of forces.
Confined aquifer	An aquifer bounded above and below by confining units of distinctly lower permeability than that of the aquifer itself. Pressure in confined aquifers is generally greater than atmospheric pressure.
Contaminant	Biological (e.g. bacterial and viral pathogens) and chemical (*see* Toxicants) introductions capable of producing an adverse response (effect) in a biological system, seriously injuring structure or function or producing death.

Term	Description
Dewatering	The lowering of static groundwater levels through complete extraction of all readily available groundwater, usually by means of pumping from one or several groundwater bores.
Dilution	The process of making a substance less concentrated by adding water. This can lower the concentrations of ions, toxins and other substances.
Drawdown	The reduction in groundwater pressure caused by extraction of groundwater from a confined formation, or the lowering of the watertable in an unconfined aquifer.
Electromagnetic	Relating to electromagnetism, which is a force described by electromagnetic fields and has innumerable physical instances, including the interaction of electrically charged particles and the interaction of uncharged magnetic force fields with electrical conductors.
Fault	A planar fracture or discontinuity in a volume of rock across which there has been significant displacement along the fractures as a result of earth movement.
Formation water	A term used largely within the petroleum industry for groundwater that occurs within petroleum or gas reservoirs.
Fracture	Any planar or curviplanar discontinuity or break in a rock mass that has formed as a result of a brittle deformation process. Joints, shear fractures, faults, microcracks, etc. are all examples of fractures.
Geologic stratum	A layer of sedimentary rock or soil with internally consistent characteristics that distinguish it from other layers. The 'stratum' is the fundamental unit in a stratigraphic column and forms the basis of the study of stratigraphy.
Geological layer	A layer of a given sample. An example is Earth itself. The crust is made up of many different geological layers, which are made up of many different minerals/substances. The layers contain important information about the history of the planet.
Groundwater	Water occurring naturally below ground level (whether in an aquifer or other low-permeability material), or water occurring at a place below ground that has been pumped, diverted or released to that place for storage. This does not include water held in underground tanks, pipes or other works.
Groundwater monitoring / observation bore	A bore installed to determine the nature and properties of subsurface groundwater conditions; provide access to groundwater for measuring level, physical and chemical properties; permit the collection of groundwater samples; and/or to conduct aquifer tests.
Hydraulic conductivity	The rate at which a fluid passes through a permeable medium.
Hydraulic fracturing	The process by which hydrocarbon (oil and gas) bearing geological formations are 'stimulated' to enhance the flow of hydrocarbons and other fluids towards the well. The process involves the injection of fluids, gas, proppant and other additives under high pressure into a geological formation to create a network of small fractures radiating outwards from the well through which the gas, and any associated water, can flow. Also known as 'fracking', 'fraccing' or 'fracture simulation'.
Hydraulic gradient	The change in hydraulic head between different locations within or between aquifers or other formations, as indicated by bores constructed in those formations.

Term	Description
Hydraulic head	The potential energy contained within groundwater as a result of elevation and pressure. It is indicated by the level to which water will rise within a bore constructed at a particular location and depth. For an unconfined aquifer, it will be largely subject to the elevation of the watertable at that location. For a confined aquifer, it is a reflection of the pressure that the groundwater is subject to and will typically manifest in a bore as a water level above the top of the confined aquifer, and in some cases above ground level.
Hydraulic pressure	The total pressure that water exerts on the materials comprising the aquifer. Also known as pore pressure.
Hydrogeology	The area of geology that deals with the distribution and movement of groundwater in the soil and rocks of Earth's crust (commonly in aquifers).
Hydrology	The study of the movement, distribution and quality of water on Earth and other planets, including the hydrologic cycle, water resources and environmental watershed sustainability.
InSAR	Satellite interferometric synthetic aperture radar: a remote-sensing technique that uses radar signals to interpolate land surface elevation changes.
Inter-aquifer leakage	Groundwater interaction between aquifers that are separated by an aquitard.
Lidar	Light detection and ranging: a remote-sensing method used to examine the surface of Earth.
Lithology	The lithology of a rock unit is a description of its physical characteristics visible at outcrop, in hand or core samples or with low magnification microscopy, such as colour, texture, grain size or composition.
Longwall mining	A method used to extract large rectangular panels of coal. The coal is progressively mined by a shearer that shaves off slices of coal from the face, under the protection of self-advancing hydraulic supports, until all the panel is fully extracted. The hydraulic supports are then removed, allowing the goaf to cave into the mined void.
Longwall mining panel	A block of solid coal whose minimum dimension (its width, equal to the face length) is typically 200–300 m in present-day Australian mines. The panel length (its maximum dimension) is generally 1–3 km. A series of panels is usually laid out side-by-side in groups of three to five.
Permeability	The measure of the ability of a rock, soil or sediment to yield or transmit a fluid. The magnitude of permeability depends largely on the porosity and the interconnectivity of pores and spaces in the ground.
Porosity	The proportion of the volume of rock consisting of pores, usually expressed as a percentage of the total rock or soil mass.
Proppant	A solid material, typically treated sand or man-made ceramic materials, designed to keep an induced hydraulic fracture open, during or following a fracturing treatment.
Radar	Radio detection and ranging: an object-detection system that uses radio waves to determine the range, altitude, direction or speed of objects.
Saturated zone	That part of Earth's crust beneath the regional watertable in which all voids, large and small, are filled with water under pressure greater than atmospheric.

Term	Description
Screen	The intake portion of a bore, which contains an open area to permit the inflow of groundwater at a particular depth interval, while preventing sediment from entering with the water.
Sediment	A naturally occurring material that is broken down by processes of weathering and erosion, and is subsequently transported by the action of wind, water or ice, and/or by the force of gravity acting on the particle itself.
Shearing	The relative, near-horizontal or low-angle movement between two sections of a rock stratum or a number of strata due to failure of the rock along a shear plane.
Slug test	A particular type of aquifer test where water is quickly added (i.e. slug test or falling head) or removed (i.e. bail test or rising head) from a groundwater well and the change in hydraulic head is monitored through time, to determine the near-well aquifer characteristics.
Stratigraphy	A branch of geology which studies rock layers (strata) and layering (stratification).
Subsidence	Usually refers to vertical displacement of a point at or below the ground surface. However, the subsidence process actually includes both vertical and horizontal displacements. These horizontal displacements, in cases where subsidence is small, can be greater than the vertical displacement. Subsidence is usually expressed in units of millimetres (mm).
Tilt	The change in the slope of the ground as a result of differential subsidence. It is calculated as the change in subsidence between two points divided by the distance between those points. Tilt is usually expressed in units of millimetres per metre (mm/m), or as a ratio of rise to run (mm:mm). A tilt of 1 mm/m is equivalent to a change in grade of 0.1 per cent.
Tiltmeter	An instrument designed to measure very small changes from the vertical level, either on the ground or in structures.
Toxicant	A chemical capable of producing an adverse response (effect) in a biological system at concentrations that might be encountered in the environment, seriously injuring structure or function or producing death. Examples include pesticides and heavy metals.
Triassic	The period of geologic time, 248 million to 206 million years ago.
Unconfined aquifer	An aquifer that has the upper surface connected to the atmosphere.
Unconsolidated sediments/materials	Sediments or materials that are not bound or hardened by mineral cement, pressure or thermal alteration.
Water quality	The physical, chemical and biological attributes of water that affect its ability to sustain environmental values.
Water quantity	A mass of water and/or discharge. It can also include aspects of the flow regime, such as timing, frequency and duration.
Watertable	The upper surface of a body of groundwater occurring in an unconfined aquifer. At the watertable, pore water pressure equals atmospheric pressure.
Well	A human-made hole in the ground, generally created by drilling, to obtain water. *See also* bore

Term	Description
Yield	The rate at which water (or other resources) can be extracted from a pumping well, typically measured in litres per second (L/s) or megalitres per day (ML/d).

1 Introduction

The Temperate Highland Peat Swamps on Sandstone (THPSS) ecological community consists of both temporary and permanent swamps developed in peat overlying Triassic Sandstone formations at high elevations, generally between 600 and 1200 m above sea level (DSEWPaC 2012a). This ecological community is largely located in the Sydney Basin in New South Wales (NSW). The THPSS is listed as an endangered ecological community under the *Environment Protection and Biodiversity Conservation Act 1999* and is also listed as endangered under the New South Wales *Threatened Species Conservation Act 1995*.

Many similar peat swamps that exist in areas below 600 m, such as the Woronora Plateau, are not included in the THPSS listing; however, where relevant, information on the Woronora Plateau swamps is considered in this report.

Collectively, the THPSS and Woronora Plateau swamps are referred to as upland peat swamps. These swamps are potentially impacted by longwall coal mining, and associated changes in the water regime, water quality, geology and topography.

This report is the first in a series of three reports focused on peat swamps and longwall coal mining that were commissioned by the Department of the Environment on the advice of the Interim Independent Expert Scientific Committee on Coal Seam Gas and Coal Mining:

- Report 1: Peat swamp ecological characteristics, sensitivities to change, and recommendations for monitoring and reporting regimes (this report)
- Report 2: Longwall mining engineering design—subsidence prediction, buffer distances and mine design options (CoA 2014a)
- Report 3: An evaluation of mitigation and remediation techniques for peat swamps impacted by longwall mining (CoA 2014b).

The objectives of this knowledge project were to:

- provide a hydrological and geological characterisation of the peat swamp communities
- model the sensitivity of the swamps to changes in the surface and groundwater flows, and changes in water quality, caused by longwall mining
- advise on the development of a monitoring and reporting approach to detect the potential impacts of longwall mining on the swamps.

Project tasks included:

- reviewing the literature on the ecology, geology, hydrogeology and hydrology of the peat swamp community. This review established three conceptual models that broadly characterise the peat swamps: headwater swamps, valley infill swamps and hanging swamps
- modelling the sensitivity of the swamps to impacts from longwall mining, using a Bayesian belief network (BBN). BBNs were developed for each conceptual model and for the community as a whole, as well as for a selection of species of flora and fauna found within the swamps. The models indicated which effect from longwall mining was likely to have the greatest impact on the peat swamps
- evaluating monitoring techniques that can be used to identify impacts on the swamps, and recommending (and scoping) monitoring and reporting approaches to be adopted by

mining proponents. The techniques primarily focus on monitoring ecological impacts on the peat swamps.

This report is intended mainly for aquatic ecosystem researchers, government agencies involved with regulation of coalmining, and mining companies whose operations may impact on peat swamps. It provides an overview and conceptualisation of upland peat swamps, and analyses the sensitivity of upland peat swamp communities and species to changes in environmental processes as a result of longwall coalmining. It also evaluates monitoring techniques, provides recommendations on monitoring programmes and methods, and identifies knowledge gaps.

2 Overview of Temperate Highland Peat Swamps on Sandstone

The Temperate Highland Peat Swamps on Sandstone (THPSS) ecological community consists of either ephemeral or permanent swamps developed in peat overlying Triassic Sandstone formations (DSEWPaC 2012a). The current *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act) listing specifies that the swamps occur at high elevations, generally between 600 and 1200 m above sea level (DSEWPaC 2012a).

2.1 Location

Swamps in the THPSS community occur in the geographic regions of the Blue Mountains, Newnes Plateau and the Southern Highlands—all within New South Wales. An additional swamp included in the EPBC listing is located at Jacksons Bog on the Victoria – New South Wales border; however, this swamp is not subject to longwall coalmining and has not been considered in this project. The swamps included in the EPBC-listed THPSS community are shown in Figure 2.1 and include (DSEWPaC 2012a):

- Blue Mountains swamps in the upper reaches of Hawkesbury River (such as Grose River and Wentworth Creek) and Nepean River (such as Bedford Creek and upper tributaries of Coxs River)
- Newnes Plateau swamps in the upper reaches of Wolgan River, Wollangambe River, Bungleboori Creek, Nine Mile Creek, Nayook Creek and Coxs River
- Southern Highlands swamps, including:
  - Butlers Swamp on the upper reaches of Nepean River
  - Gallaghers Swamp and Rock Arch Swamp on the upper reaches of Avon River
  - Paddys River swamps, including Jumping Rock Swamp, Hanging Rock Swamp, Mundego Swamp, Long Swamp and Stingray Swamp
  - North Pole Swamp and Stockyard Swamp on the upper reaches of Dudewaugh Creek
  - Wildes Meadow Swamp on the upper reaches of Shoalhaven River
  - Wingecarribee Swamp on the upper reaches of Wingecarribee River.

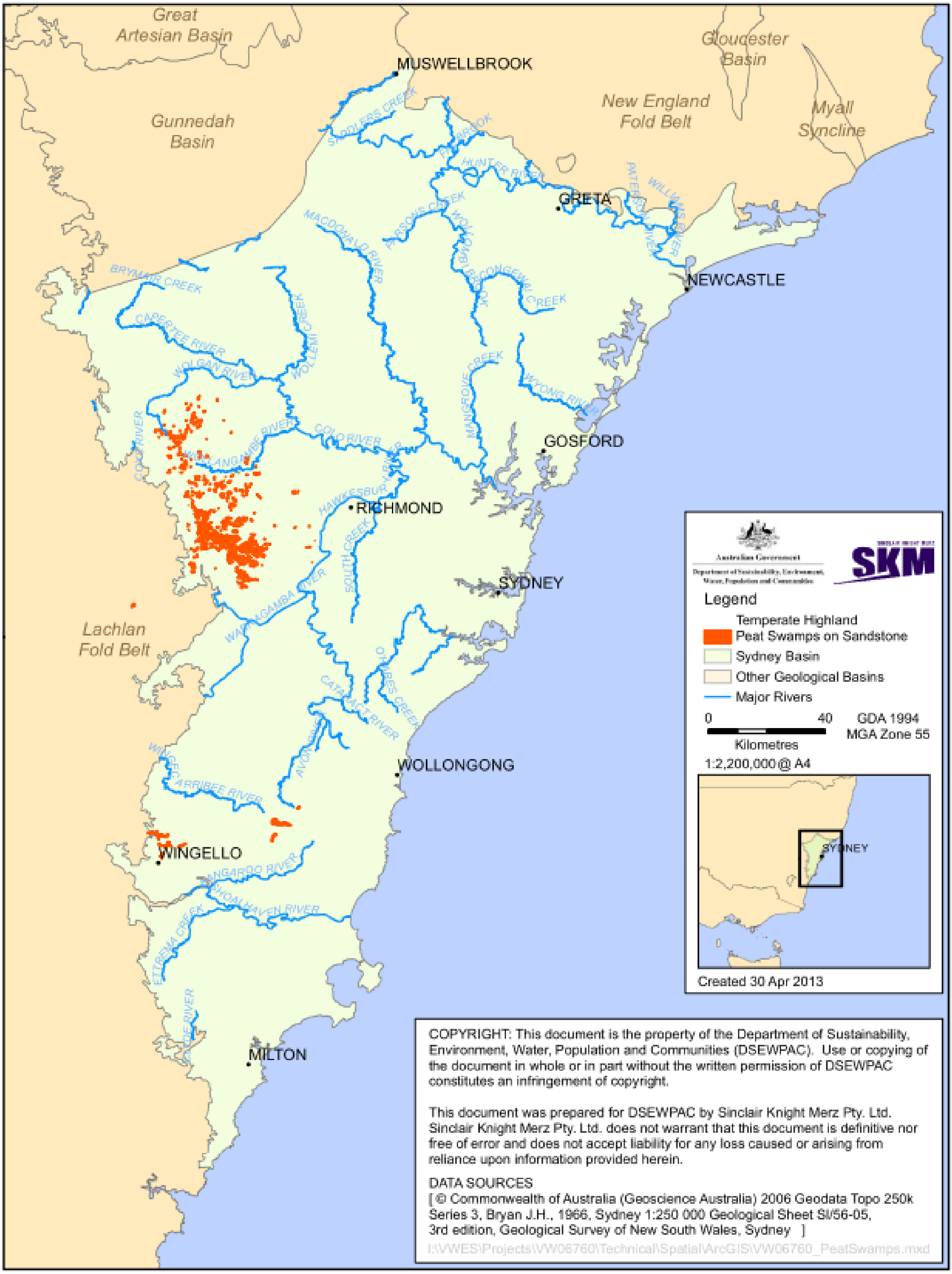
Many similar peat swamps that exist in areas below 600 m, such as the Woronora Plateau, are not included in the THPSS listing; however, where relevant, information on these swamps is considered in this project (see note 1 at the end of the chapter) .

Tomkins and Humphreys (2006) describe swamps on the Woronora Plateau as either ‘headwater swamps’ or ‘valley infill swamps’. Headwater swamps make up the majority of all swamps in the Southern Coalfields and occur near catchment divides where topographic grades are shallow. They form over low-permeability substrates of sandstone formations or clay horizons (Young 1986). Approximately 6444 ha of this type of swamp have been mapped on the Woronora Plateau (NPWS 2003). Valley infill swamps are less common and infill the more dissected valleys of second- or third-order streams. Examples of valley infill swamps are Flatrock Swamp on Waratah Rivulet, swamps 18 and 19 on Native Dog Creek, and Martins Swamp (Tomkins & Humphreys 2006). Some of the larger swamps could be considered headwater swamps in one part, and valley infill swamps in another (NSW PAC

2009). Hanging swamps occur on the sides or cliffs of steep valleys where groundwater discharges to the surface. They have been mapped most extensively in the Blue Mountains and Newnes Plateau, but have also been identified in the Bargo and Cataract gorges on the Woronora Plateau (NSW DP 2008).

2.2 Geology

In the Blue Mountains and Newnes Plateau the swamps are associated with the Narrabeen Group (predominantly the Banks Wall Sandstone) and the Hawkesbury Sandstone, and form in gently sloping headwater valleys (Keith & Benson 1988; Benson & Keith 1990); open drainage lines in footslopes, broad valley floors and alluvial flats (NSW DEC 2006); gully heads and open depressions on ridgetops (Holland et al. 1992; BMCC 2005; NSW DEC 2006). They also occur on steep valley sides at the interface between the sandstones of the Narrabeen Group and underlying lower permeability claystone layers (Keith & Benson 1988; Holland et al. 1992; BMCC 2005) as ‘hanging swamps’. In the Southern Highlands the swamps overlie the Triassic Hawkesbury Sandstone (Stricker & Wall 1994; Winning & Brown 1994, Stricker & Stroinovsky 1995). Figure 2.2 shows the swamps and their underlying geology.



Note: Similar swamps occur below 600 m (e.g. on the Woronora Plateau) that are not currently part of the EPBC-listed community and are not shown.

Figure 2.1 Temperate Highland Peat Swamps on Sandstone ecological community occurs in the Blue Mountains, Newnes Plateau and Southern Highlands.

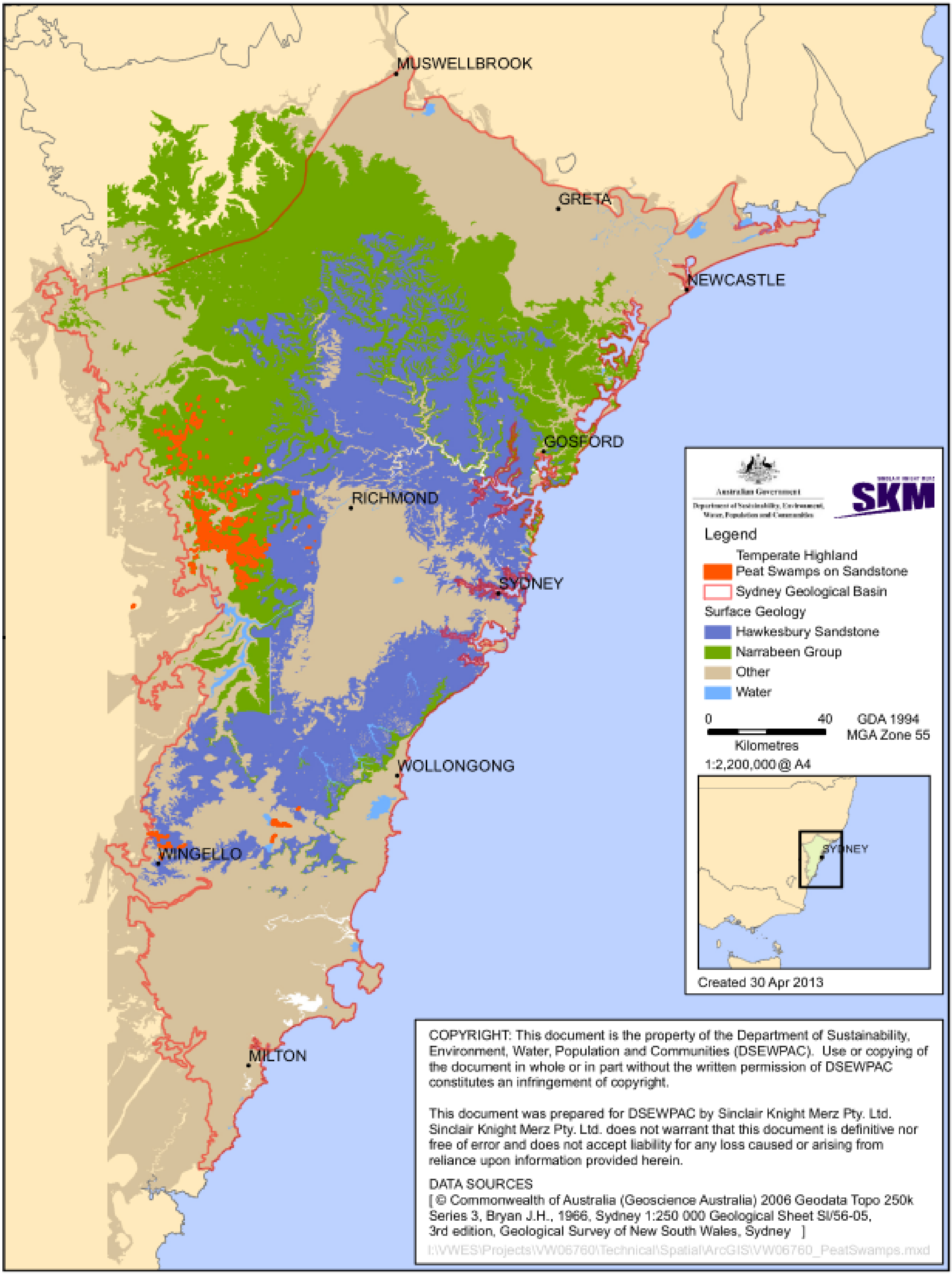


Figure 2.2 Temperate Highland Peat Swamps on Sandstone ecological community occurs on Hawkesbury Sandstone in the Southern Highlands and Banks Wall Sandstone (Narrabeen Group) in the Blue Mountains and Newnes Plateau.

2.3 Climate of the Sydney Basin

The peat swamps occur in the temperate climate zone, which is described as having a warm summer and cool winter. The average daily maximum temperate is between 15°C and 24°C, with variation largely depending on elevation. Average annual rainfall is between 800 and 1600 mm and is not strongly seasonal, although winter and spring tend to be slightly drier than summer and autumn.[1] On the Woronora Plateau, highest rainfall occurs close to the coast on the Illawarra Escarpment (Tomkins & Humphreys 2006), and generally in more elevated areas.

2.4 Longwall mining in the Sydney Basin

The peat swamps overlie the Southern and Western coalfields in the Sydney Basin. In 2012, there were eight operating longwall mines in the Southern Coalfield and three in the Western Coalfield, in areas where peat swamps may occur at the surface. These mines collectively produce about 60 million tonnes of coal per year and mine beneath approximately 10 km^2 of land each year. Table 2.1 lists the operating coalmines in the Southern and Western coalfields.

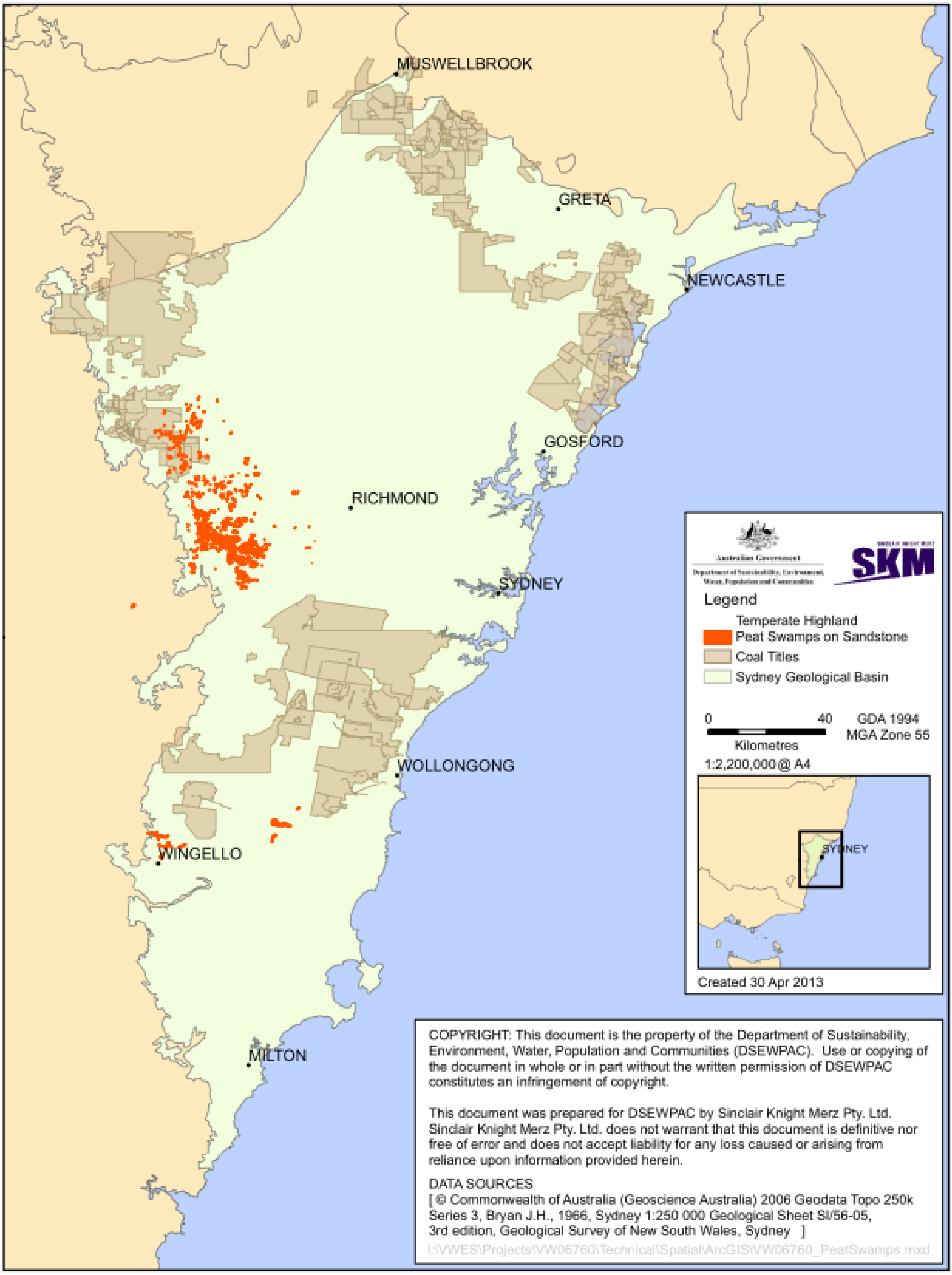
Figure 2.3 shows the coal titles and indicates the part of the peat swamp community that is at risk of being undermined and therefore more at risk of subsidence impacts. Most swamps in the Southern Highlands (and Woronora Plateau) are not shown in Figure 2.3 since they are below 600 m elevation and are therefore not listed as part of the EPBC community. However, many swamps exist in this area and are threatened by impacts associated with the longwall mines.

[1] www.bom.gov.au/climate

Table 2.1 Longwall coal mines and operators in the Southern and Western Coalfields, New South Wales.

Southern Coalfield[a]	**Operator**
Metropolitan Colliery	Helensburg Coal Pty Ltd, a subsidiary of Peabody Energy Australia Coal Limited
West Cliff Colliery	Illawarra Coal Holdings, a subsidiary of BHP Billiton Group
Appin and Appin West Colliery	Illawarra Coal Holdings, a subsidiary of BHP Billiton Group (formerly called Tower Colliery)
Dendrobium Colliery	Illawarra Coal Holdings, a subsidiary of BHP Billiton Group
NRE No 1 Colliery	Gujarat NRE Australia Pty Ltd (formerly called Bellpac, South Bulli, Bellambi West Colliery)
Wongawilli Colliery	Gujarat NRE Australia Pty Ltd (before 2007, owned by Illawarra Coal Holdings and called Elouera Colliery)
Tahmoor Colliery	Xstrata Coal (NSW) Pty Ltd (before 2007, owned by Centennial Coal Company Limited)
Berrima Colliery	Centennial Coal Company Limited
Western Coalfield[b]	
Angus Place Colliery	Centennial Coal Company Limited
Springvale Colliery	Centennial Coal Company Limited
Baal Bone Colliery	Xstrata Coal (NSW) Pty Ltd. Mining ceased in 2011[c]
Ulan West Colliery	Xstrata Coal (NSW) Pty Ltd. Second longwall mine opening in 2014[c]
Invincible Colliery	Coalpac. Currently operating as a bord and pillar/open-cut mine, but was a longwall operation before 1988[d]

a Data from NSW DP 2008
b Data from http://theland.farmonline.com.au/news/state/agribusiness-and-general/general/lithgow-cliffs-crumbled-due-to-coalmining/1813260.aspx
c www.xstratacoal.com/EN
d http://cetresources.com/operations/invincible-colliery
Source: Adapted from NSW DP 2008



Note: Many similar swamps occur below 600 m that are not part of the EPBC community and are not shown.

Figure 2.3 Coalmining leases in the Sydney Basin.

Peer review comments on Chapter 2

1. In December 2010, the Threatened Species Scientific Committee requested that the then Australian Government Department of Sustainability, Environment, Water, Population and Communities review the listed ecological community Temperate Highland Peat Swamps on Sandstone with the view that it could be split into two separate ecological communities, one of which was the Upland Peat Swamps of the Sydney Sandstone Basin, later renamed Peaty Upland Swamps on Sandstone in the Sydney Basin. The revision concluded that the proposed changes would be confusing and would not achieve the desired protection outcomes. Subsequently, the decision was made to keep the original Temperate Highland Peat Swamps on Sandstone ecological community without change and examine the recently listed New South Wales Coastal Upland Swamp in the Sydney Basin Bioregion to assess whether it would be appropriate to list a similar or the same ecological community under the national environmental law.

   The assessment was placed on the 2013 Finalised Priority Assessment List by the federal Minister for the Environment. Guided by expert technical input, an assessment of eligibility for listing by the Threatened Species Scientific Committee has progressed in the form of a draft description of the ecological community.

   Early in 2014, the Department of the Environment undertook a public consultation on the assessment of the Coastal Upland Swamps in the Sydney Basin Bioregion as a potentially threatened ecological community under Australia's national environmental law, the *Environment Protection and Biodiversity Conservation Act 1999* (the EPBC Act). It was proposed to assess the community for listing as 'endangered'. At the time of writing, the outcome of the proposed listing was undecided.

The following peat swamp geomorphological description was provided by NSW state agencies, which adds to the overview of Temperate Highland Peat Swamps on Sandstone presented in Chapter 2. References in the text below are listed in the References, under 'Additional references provided by peer reviewers'.

Context

The Blue Mountains region is defined by a monoclinal uplift, which has elevated the plateaus of the region by a maximum of 800 m above its previous level. This has significantly affected the geomorphology and fragility of the rivers that cross it. Rivers have been dramatically influenced by overall changes to long profile gradients and enhanced incision into the underlying Triassic Sandstone sequence. The formation of rivers and associated wetlands on the Newnes, Kanangra and Woronora plateaus, their morphology and hydrological and energy transmission characteristics require a thorough understanding of the nature of the controls imposed on them, and the likelihood of change following catchment disturbance, including mining-induced subsidence.

Rivers in the upland areas of the Blue Mountains are often planform controlled by geological structures, including faults, jointing networks and bedding plane outcropping. This has created complex relationships between surface and shallow ground water flow, which may be difficult to differentiate (McKibben & Smith 2000). The combination of local geological controls on river planform and the steep escarpments to all sides of the plateaus has created three general river and wetlands systems:

- 'diffuse' poorly defined flow pathways with significant deposition of sand and organic loams on the broad plateau crests—the sites of the many identified highland peat swamps
- canyon to gorge river valleys with minimal wetland development, though frequently possessing rare or endangered oligofaunal and troglofaunal communities
- transitional rivers between upland discontinuous flowpath systems and canyon/gorge rivers, which exhibit well-defined channels, but may be depositional zones because of bedrock outcrop outlet controls—the sites for a significant number of highland peat swamps.

The geomorphic processes of peat swamp development are critical to maintaining their integrity and function. The upland swamp systems are categorised as having high geomorphic fragility, due to the unchannelised fill surfaces of the wetland system, and the unconsolidated, high sand content fill overlying sandy peat (Freidman 2011 in Fryirs et al. 2012).

Urbanisation on the Blue Mountains plateaus has altered both channel form and flow response in the catchments of a number of highland swamps (Fryirs et al. 2012). Mining-induced subsidence has been studied over several decades following severe impacts caused by mining-induced subsidence (Tompkins & Humphrey 2006; Galvin & Associates 2005).

Numerous mechanisms may cause disturbance and instability in upland swamps. Although mining-induced subsidence is identified as a key threatening process to upland swamp integrity (NSW Scientific Committee 2005), other catchment activities may cause changes in catchment and swamp hydrology. An example is the effect of urbanisation on the network of upland swamps south of the Grose River gorge, parallel to the Great Western Highway. Concentration of stormwater flow and track disturbance and flow channelisation along tracks has resulted in approximately 50 per cent of 47 surveyed upland swamps changing geomorphic classification from 'valley fill' to 'channelised fill' rivers (Fryirs et al. 2012). Consequential deterioration of these wetlands has resulted in their drying out and loss of saturation dependent vegetation, with incursions of dryland and exotic vegetation (Friedman 2011 in Fryirs et al. 2012).

Existing damage to upland swamps

A number of rockbar outlet-controlled swamps in transitional rivers have been significantly damaged or degraded as a result of urbanisation and mining-induced subsidence. The process of formation of these upland and transitional river reach swamps has been studied more fully to impacted swamps on the Woronora Plateau, such as Drillhole Swamp (Young 1982; Gibbins 2003), Flat Rock Swamp (Mills & Huuskes 2004) and Swamp 18 (Biosis Research 2001; Earth Tech 2005). The assessment of the severity of impact to these wetlands was often impeded by a lack of adequate pre-subsidence monitoring data (Paterson 2004).

The two mixed heath/valley fill swamps (Drillhole and Flat Rock swamps), which were initially studied to determine relative and absolute consequential effects of mining-induced subsidence effects, have not recovered from the combination of drainage, channelisation and fire. More recent studies (Keith et al 2006; Tomkins and Humphreys 2006; NSW Government 2008) have confirmed the severity and decadal duration of such impacting change on shallow valley fill water storage and erosive energy on highland swamps.

Fryirs et al. (2012) identified 23 of 47 surveyed swamps between Lawson and Medlow Bath had experienced varying degrees of degradation from concentrated stormwater run-off from nearby urban areas. The most common effect was channelisation through relatively featureless valley fill surfaces and consequential drainage of saturated loam and peat fill. This consequence has been observed in the Southern Coalfield (Mills & Huuskes 2004; Young 1982), and has been linked to increased bushfire risk through the dried out vegetative mats and underlying peat beds following channelisation.

It is clear both direct (e.g. mining subsidence base rock fracturing, induced drainage, profile gradient change) and indirect (e.g. catchment stormwater concentration) effects may have a significant influence on the geomorphic stability and integrity of upland swamps. This includes measured incision and channelisation to more than 90 per cent of the valley fill section (Tomkins & Humphreys 2006; Freidman 2011), with consequential increased risk of fire extending through and destroying both living vegetation associations and exposed peat.

3 Peat swamp conceptualisation

An accurate conceptualisation of the environmental relationships that control the presence of the peat swamps is required to assess sensitivity of the swamps to change, and also to recommend the most appropriate monitoring regimes. This section presents the knowledge available on peat swamps, based on the literature reviewed. Three conceptual models have been outlined in the literature:

- headwater swamps—formed near catchment divides where topographic gradients are shallow
- valley infill swamps—occur in steeper topographies filling the valley of incised second- or third-order streams
- hanging swamps—occur on steep valley sides where there is groundwater seepage.

Tomkins and Humphries (2006), the Southern Coalfield review (NSW DP 2008), the Metropolitan Coal project review report (NSW PAC 2009) and the Bulli Seam Operations PAC report (NSW PAC 2010) characterise the swamps using these conceptual models. The primary characteristics of each model are outlined below, followed by a more detailed analysis of the geological, hydrological and ecological characteristics of the swamps. This information develops an understanding of the characteristics and environmental requirements of the peat swamps, and thus of the risk to the peat swamps when the environment is altered.

3.1 Conceptual models

3.1.1 Headwater swamps

Headwater swamps occur close to catchment divides at the headwaters of streams where the topographic gradient is low. They occur throughout the Sydney Basin and are the dominant swamp type on the Woronora Plateau (NSW DP 2008). Because of their position high in the landscape and the relatively flat terrain in which they occur, groundwater connection is generally limited. The water regime is dominated by rainfall and surface water run-off. The characteristics of headwater swamps are shown diagrammatically in Figure 3.1 and summarised below.

3.1.1.1 Geology/Substrate

- Sandy/clayey peat overlying low-permeability sandstone (Tomkins & Humphreys 2006).
- Sedimentation of sand, clay and peat within the swamp is controlled by water depth in the swamp, run-off from local catchment and vegetation type present (Whinam & Chilcott 2002; Price et al. 2003; Nanson 2006).
- Peat thickness is variable. Median depth in the Southern Highlands is 40 cm (Whinam & Chilcott 2002) but can be up to 10 m, although this type of depth only relates to one Wingecarribee swamp (Kodela & Hope 1992; Winning & Brown 1994; Stricker & Stroinovsky 1995).

3.1.1.2 Water regime

- Surface of the swamps can be either permanently or ephemerally wet (DSEWPaC 2012a).

- Water quality within the swamps is controlled by catchment run-off.
- The dominant water source of the swamp is recharge through rainfall and run-off (both overland flow and from headwater streams) (NSW PAC 2009).
- Water flows through the swamps either as sheet flow along the surface of the peat, through the peat sediments themselves or through channels that are normally discontinuous in the peat within the peat.

3.1.1.3 Groundwater connection

- Swamps are often perched above the watertable, especially in the Upper Nepean area; however, they may be connected to a shallow perched aquifer within the Banks Wall Sandstone (Blue Mountains/Newnes Plateau) or the Hawkesbury Sandstone (Blue Mountains/Newnes Plateau, Southern Highlands, Woronora Plateau) (NSW PAC 2009; NSW DP 2008; Young 2007).
- Recharge to the perched source aquifers is through recently infiltrated rainfall (NSW DP 2008).
- Groundwater can discharge to the swamps through:
    - groundwater flow along fractures, joints or bedding planes that intersect the peat swamp
    - sandstone layers that intersect the peat swamp, although this is less common than along fractures and joints (Coffey 2008).
- Groundwater that interacts with the peat swamps is from a local flow system, and therefore has short flow paths and residence times.
- Where groundwater discharges to the swamps, water quality is expected to be fresh because of relatively short residence times.
- If a connection between groundwater and a swamp exists, the connection is most likely to be ephemeral because it relies on the presence of a perched aquifer, which is most likely to be present after rainfall.

3.1.1.4 Flora

- In the Blue Mountains/Newnes Plateau:
    - swamps support shrublands and heathlands, which are generally dominated by shrubs and sedges
    - common species include baeckeas (*Baeckea linifolia* and *B. utilis*), swamp heath (*Epacris paludosa*) and razor sedge (*Lepidosperma limicola*). Deane’s boronia (*Boronia deanei*) is a threatened species that has been observed in headwater swamps (DSEWPaC 2012a
- In the Southern Highlands, cyperoid heath, tea-tree thicket, banksia thicket, restoid heath, sedgeland and fringing eucalypt woodland have been commonly observed in headwater swamps (HCPL Coal 2008).

3.1.1.5 Threats to swamps from longwall mining

- Fracturing and tilting of underlying sandstone associated with subsidence.
- Less vulnerable to subsidence than valley infill swamps, but this may be just because the impacts have not been observed or monitored (NSW DP 2008).

- Because they occur high in the catchment, headwater swamps are not considered to be at risk of water quality impacts arising from discharge of mine waste water. Mine waste water is usually discharged further downstream, at lower elevations.

3.1.2 Valley infill swamps

Valley infill swamps occur further down the catchment than headwater swamps, in the steeper terrain of incised valleys associated with second- or third-order streams. They occur across the Sydney Basin, although they are not as common as headwater swamps, at least on the Woronora Plateau (NSW PAC 2009). The steeper incision into the underlying sandstones means the swamps are more likely to intersect water-bearing layers within the horizontally bedded sandstone. The water regime for valley infill swamps therefore combines rainfall and surface water run-off, as well as groundwater inputs. The characteristics of valley infill swamps are shown diagrammatically in Figure 3.1 and summarised below.

3.1.2.1 Geology/substrate

- Sandy/clayey peat overlying low-permeability sandstone (Tomkins & Humphreys 2006).
- Sedimentation of sand, clay and peat within the swamp is controlled by water depth in the swamp, run-off from local catchment and the vegetation type present in the swamp (Whinam & Chilcott 2002; Price et al. 2003; Nanson 2006).
- Peat thickness is variable. Median depth in the Southern Highlands is 40 cm (Whinam & Chilcott 2002), but can be up to 10 m (Kodela & Hope 1992; Winning & Brown 1994; Stricker & Stroinovsky 1995).

3.1.2.2 Water regime

- The swamp surface can be either permanently or ephemerally wet (DSEWPaC 2012a).
- Water quality within the swamps is variable, and is controlled by a combination of rainfall, run-off and groundwater quality.
- The swamp is recharged through a combination of groundwater discharge from perched or regional sandstone aquifers, rainfall and run-off (NSW PAC 2009; NSW DP 2008).
- Water flows through the swamps either as sheet flow along the surface of the peat, through the peat or through channels within the peat. These channels control the water level within the peat swamps (Nanson 2006; A Young 2010, pers. comm., October).

3.1.2.3 Groundwater connection

- Source aquifers are perched or are sometimes (less commonly) regional sandstone aquifers of Banks Wall Sandstone (Blue Mountains/Newnes Plateau) or Hawkesbury Sandstone (Blue Mountains/Newnes Plateau, Southern Highlands, Woronora Plateau) (NSW PAC 2009; NSW DP 2008).
- Recharge to the source aquifers is through infiltrated rainfall (NSW DP 2008).
- Groundwater discharge to the swamps is through either:
    - groundwater movement along fractures, joints or bedding planes that intersect the peat swamp
    - to a lesser extent, the lower permeability sandstone layers that intersect the peat swamp.
- Depending on location in the landscape, the groundwater flow system could be local, intermediate or regional.

- Connection between aquifer and swamp is either permanent (more likely where the regional aquifer is the groundwater source) or ephemeral (more likely where perched aquifers are the groundwater source).
- Groundwater quality is variable, depending on residence time within the aquifer.

3.1.2.4 Flora

- In the Blue Mountains:
  - supports closed sedgeland communities with occasional shrubs (DSEWPaC 2012a)
  - common species include spreading rope rush (*Empodisma minus*), button grass (*Gymnoschoenus sphaerocephalus*), razor sedge (*Lepidosperma limicola*) and woolly tea-tree (*Leptospermum lanigerum*) (DSEWPaC 2012a).
- In the Southern Highlands, cyperoid heath and tea-tree thicket are common vegetation types (HCPL Coal 2008).

3.1.2.5 Threats to swamps from longwall mining

- Valley closure, upsidence and related fracturing and tilting of underlying sandstone.
- More vulnerable to subsidence than headwater swamps since they occur in steeper terrain where natural stresses are higher (NSW DP 2008).

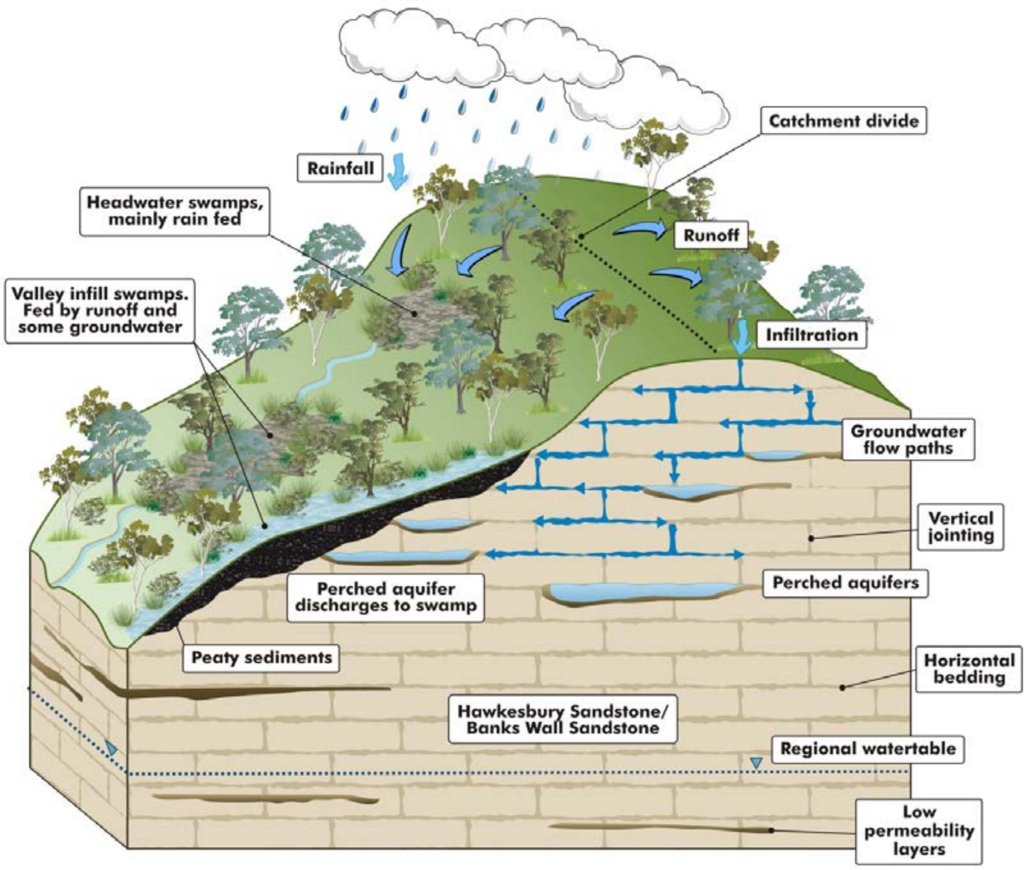


Figure 3.1 Conceptual model block diagram describing headwater and valley infill swamps.

3.1.3 Hanging swamps

Hanging swamps occur on steep valley sides or cliffs, and are predominantly reliant on groundwater discharge that seeps out along bedding planes and low-permeability layers in the sandstone. They occur most famously in the Blue Mountains and Newnes Plateau, but have also been identified in the Bargo and Cataract gorges on the Woronora Plateau (NSW DP 2008). The characteristics of hanging swamps are shown diagrammatically in Figure 3.2 and summarised below.

3.1.3.1 Geology/substrate

- Swamps occur at the interface between higher and lower permeability sandstone layers (Keith & Benson 1988; Holland et al. 1992; BMCC 2005).
- Sediment and peat deposition is minimal due to the steep topography, and is limited to sediment caught within vegetation roots (DSEWPaC 2005).

3.1.3.2 Water regime

- Swamps can be either permanently or ephemerally wet (DSEWPaC 2012a)
- Water quality is similar to local groundwater quality (and expected to be fresh).
- The dominant water source for the swamps is groundwater, which seeps to the surface at cliff faces, above lower permeability sedimentary layers.

3.1.3.3 Hydrogeology

- Source aquifers are the shallow, higher permeability parts of the Banks Wall Sandstone in the Blue Mountains/Newnes Plateau; or shallow, higher permeability parts of the Hawkesbury Sandstone.
- Recharge to the source aquifers is through recently infiltrated rainfall.
- Groundwater discharge to the swamps is caused by the presence of low-permeability layers within the aquifer forcing water sideways to seep out of the cliff face (Keith & Benson 1988). Groundwater contributions may be from perched aquifers, or as recently infiltrated water that flows along cracks and joints before discharging to the swamp.
- Groundwater flow system is local, and groundwater quality is expected to be fresh due to relatively short flow paths (i.e. less than about 10 km) and residence times in the aquifer.
- Connection between aquifer and swamp is either permanent or ephemeral and occurs after rainfall.

3.1.3.4 Flora

- In the Blue Mountains/Newnes Plateau:
  - open heath vegetation communities are dominated by shrubs and sedges (Environment Australia 2001)
  - common species include coral heath (*Epacris microphylla*), blunt-leaf heath (*Epacris obtusifolia*) and pink tea-tree (*Leptospermum squarrosum*) (Environment Australia 2001).
- No information exists on vegetation communities of hanging swamps in the Southern Highlands or Woronora Plateau.

3.1.3.5 Threats to swamps from longwall mining

- Valley closure, cliff collapse and related fracturing of sandstone.
- Hanging swamps are expected to be more vulnerable to subsidence impacts than headwater and valley infill swamps, due to their location in steep topography where natural stresses are highest.

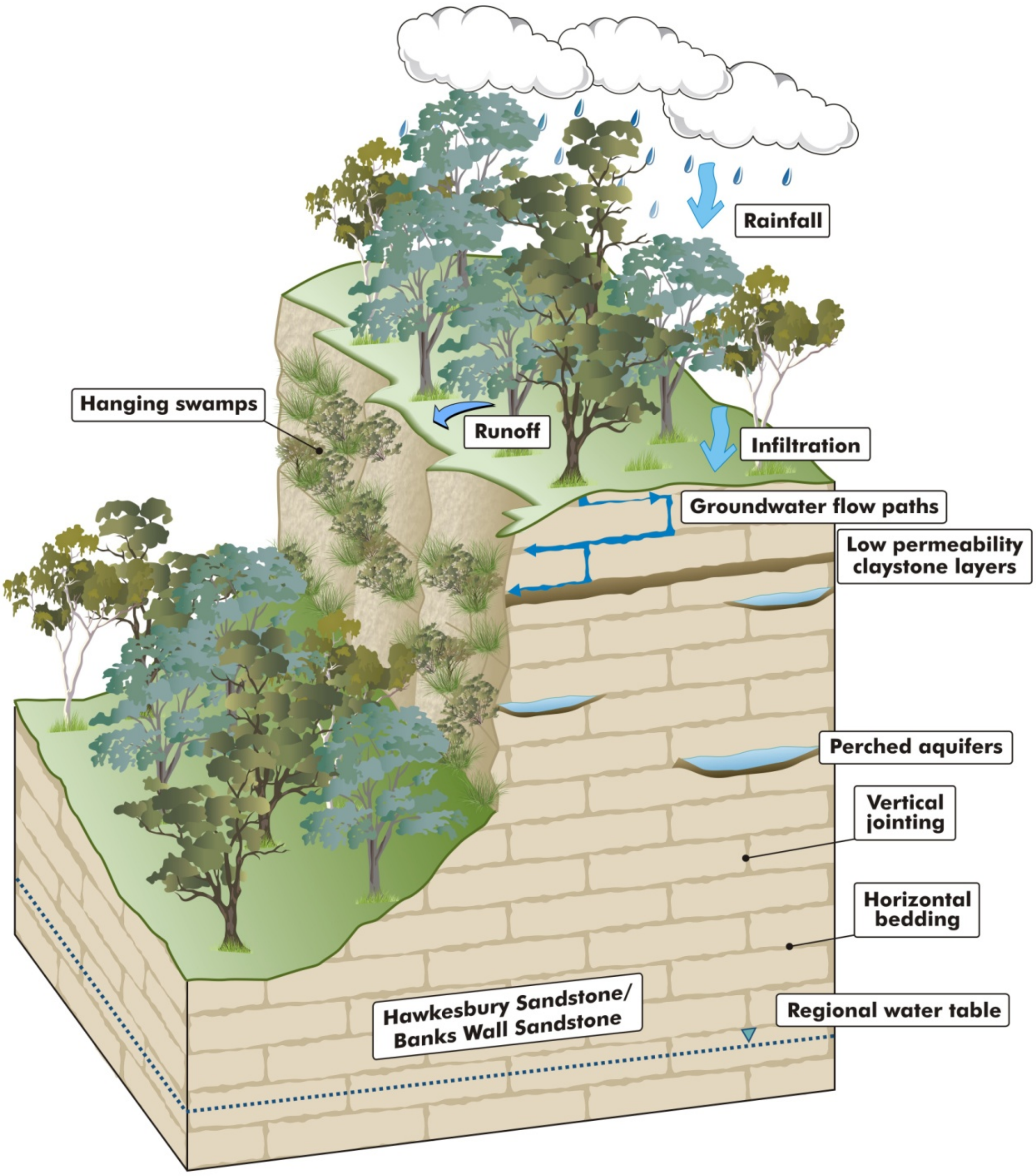


Figure 3.2 Conceptual model block diagram describing hanging swamps.

3.1.4 Summary

There are no categorical differences in the peat and geology between headwater swamps and valley infill swamps. Peat composition and depth varies significantly across the Sydney Basin and between all swamp types. Underlying geology is consistent between both swamp types, forming a low-permeability layer that controls flow between the swamp and the sandstone. For hanging swamps, peat thickness is significantly less than the other two swamp types. The sandstone may also be more permeable than for headwater and valley infill swamps, since it provides the dominant water source for the hanging swamps via flow through fractures, joints and bedding planes.

No significant differences were noted in the water regime between headwater and valley infill swamps: they can both range from permanently to ephemerally wet, and contain channels through which water flows and controls the water level within the swamps. The higher gradient of valley infill swamps may cause water to flow through at a higher velocity.

A fundamental difference between the swamp types is the varying topographical locations and the resulting differences in potential for connection to groundwater. Headwater swamps are unlikely to be connected to groundwater because they occur in flat terrain in elevated topographies, where the regional groundwater is deep and perched aquifers are unlikely to be intersected by the swamp. The dominant water source for headwater swamps is therefore rainfall and surface run-off. In contrast, valley infill swamps occur in incised topographies and intersect more of the horizontal layers in the underlying sandstone, resulting in a greater likelihood of intersecting perched or regional aquifers. Therefore, rainfall, surface run-off and groundwater discharge all contribute to valley infill swamps. Hanging swamps are more reliant on groundwater than either of the other two swamp types, since they occur on steep valley sides or cliffs where groundwater discharges directly to the surface. Because of their elevated location, perched aquifers (rather than the regional aquifer) are likely to be the primary water source for hanging swamps.

There are an estimated 1050 swamps in the community listed under the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act) and so the three conceptual models described are broad categorisations of the swamps. In reality, a single swamp may be best described by a combination of the conceptual models, particularly as larger swamps can grade from one type to another. The ecological characterisation is especially difficult to constrain into the three conceptual models presented here, since the vegetation assemblage at a given swamp is strongly influenced by wetness, elevation and topography, and these factors can vary significantly over short distances. For example, valley infill swamps near the Ulladulla escarpment are generally wetter and are therefore likely to contain a different vegetation assemblage than valley infill swamps further west.

Despite the inherent variability within the swamp community, the conceptual models described in this section provide important distinctions between swamps that can influence a swamp's susceptibility to impacts from longwall mining. These generalised conceptual models are supported by the literature and provide the necessary understanding of swamp function to build a Bayesian belief network model for the swamps (see note 1 at the end of the chapter).

3.2 Swamp stratigraphy

The Temperate Highland Peat Swamps on Sandstone (THPSS) ecological community consists of swamps developed in peat overlying Triassic Sandstone formations. The swamps have developed since the last glacial event about 16 000 years ago. This section discusses the stratigraphy of the swamp sediments that overlie the sandstones, the development of the peat substrate and subsequent erosion.

3.2.1 Swamp sedimentation

Peat swamps consists of varying depths of sediment, including clay, sand and peat overlying the Hawkesbury Sandstone in the Southern Highlands and eastern Blue Mountains, or the Banks Wall Sandstone in the western Blue Mountains – Newnes Plateau areas. The relative proportion of peat, clay and sand in the swamp substrate depends on the local catchment. Since erosion of the local catchment controls the sedimentation in the peat swamps, the local sandstone composition, level of weathering, catchment run-off and vegetation cover will influence the sediments found in the peat swamp.

A mechanism for the development of peat swamps is the obstruction of drainage lines by tree trunks, bark or rocks preventing water from flowing down the valley and causing sediment to accumulate (Tomkins & Humphreys 2006). Erosion from valley sides then deposited sediment between the cobbles. This sedimentation was encouraged by vegetation growth within the swamp as a result of a warmer climate, and had the effect of trapping more clay within root zones. Saturation of the valley infill, relatively constant watertables and reduced oxygen conditions in the substrate then resulted in the development of a peat horizon (Price et al. 2003; Nanson 2009). The depositional history described by Nanson (2009) was the result of studies on peat swamps (Edwards and Polblue swamps) in the Barrington Tops National Park north-west of Newcastle. Although these peat swamps are not part of the EPBC THPSS listing (they are located over basalt), they exist in a similar climate and may have experienced a similar depositional history and therefore a similar sedimentary layering and hydrology (see note 2 at the end of the chapter).

The description of sedimentary layering in peat swamps in the Woronora Plateau in Tomkins and Humphreys (2006) identifies a sandy peat profile, while further north in the Barrington Tops National Park, Nanson (2009) reports a clay-dominated peaty profile. This emphasises that the composition of the swamp substrate is variable, and contains varying proportions of sand, clay and peat. The peaty-type soils are grey to black, acidic and reducing, with a high content of organic matter and a sandy or loamy texture (DSEWPaC 2012a). Fires and subsequent heavy rainfall have also caused charcoal to accumulate within the peat layers. The peats are poorly drained and generally remain waterlogged, either permanently or intermittently (DSEWPaC 2012a).

The stratigraphy of Wingecarribee Swamp in the Southern Highlands is unique because the peat swamp is separated from the Hawkesbury Sandstone by a layer of Wianamatta Shale. Overlying the shale is high plasticity clay containing minor quartz fragments, in turn overlain by a layer of peat (Coffey 2004).

The thickness of the peat is highly variable even within individual swamps and depends on the topographic setting of the swamp; in particular, the degree of incision of the valley. As sedimentation is controlled by the water level within the swamps, peat thickness is also determined by the depth and temporal nature of waterlogging. Peat thickness as little as 1 cm has been reported at swamps in the Blue Mountains, and a median depth of 40 cm has been reported for swamps in the Southern Highlands (Whinam & Chilcott 2002). Peat thickness in Wingecarribee Swamp can be up to 10 m (Kodela & Hope 1992; Winning & Brown 1994; Stricker & Stroinovsky 1995).

The swamps act as a sediment store in the catchment, trapping sediment that would otherwise be transported downstream. They are therefore recognised as being an important filter for the downstream catchment.

This description of the substrate of peat swamps applies to topographic locations that allow the build-up of sediments, such as valley floors or stream headwaters. However, hanging swamps are also included in the EPBC listing, and these occur on steep valley sides where there is limited opportunity for sediments to accumulate. Hanging swamp ecosystems occur where groundwater is discharged at valley sides, caused by groundwater flow along a lower permeability claystone within the Triassic Sandstone. The layering described above therefore does not apply to these unique ecosystems. For hanging swamps, the presence of peat is likely to be limited to the amount that can be trapped by, and accumulate in, roots of the hanging swamp vegetation.

3.2.2 Erosion

'Gullying' refers to the erosion of peat swamps along drainage lines, where larger channels are carved into the surface of the swamp. This can alter the hydrological regime of the swamp, lowering the watertable and allowing erosion of the swamp at the surface and through scouring along the gullies or channels. Tomkins and Humphreys (2006) attempted to establish a link between the development of gullies in three peat swamps on the Woronora Plateau and catalysts of erosion, including severe rainfall events, fire, drought and subsidence caused by coalmining. Gullying could not be attributed to a single cause in these swamps, and was concluded to be initiated by a combination of events—in particular, fires followed by severe rainfall events (Young 1982). Gullying is therefore thought to be a normal part of the depositional and erosional cycle of peat swamps (Tomkins & Humphreys 2006).

Dewatering of the swamps can result in subsidence of the swamp surface as the peat blocks dry out, compress and fissure. For example, after the collapse of Wingecarribee Swamp, dewatering caused peat blocks to reduce in size from around 6 m to 2 m (Hope 2003). This caused loss of wetland species in the swamp (Sainty & Associates 2003). As the peat destabilises it can also be transported downstream as peat balls that have cracked away from the dried peat substrate (A Young 2010, pers. comm., October).

3.3 Swamp hydrology

Peat swamps rely on permanent or regular waterlogging and are very sensitive to changes in hydrology (DSEWPaC 2012a). They are characterised by a water level that remains close to, or above, the surface of the swamp, at least for several months of the year. Water levels within the swamp sediments are usually closer to the surface, along the axis of the valley, and fluctuate in response to rainfall and evaporation (Holland et al. 1992).

Flow across the surface of the swamp is controlled by the dense stems of vegetation, resulting in a wide, shallow sheet flow, which distributes sediment relatively evenly across the surface of the swamp (A Young 2010, pers. comm., October).

The peat swamps can be incised by channels that are usually narrow and deep. These channels flow perennially and are often close to full along their entire length (Nanson 2006). Movement of water through the swamp is largely controlled by these channels, since the level of water in the channels also controls the watertable height within the peat (Nanson 2006). The swamp sediments release water to the channels in a regular manner, as demonstrated by the diurnal variations in channel flow identified by Nanson (2006). Natural hydrological responses within the swamps occur hourly and daily in response to changing weather conditions (Holland et al. 1992).

The consistency of flow and the height of water within swamp channels are controlled largely by the extent of channelling within the peat swamps, and this determines the depth of peat development (Nanson 2009). That is, the natural extent and morphology of the channels maintain a relatively constant water level, which allows peat to accumulate. An increase in channel size or extent allows water to flow out of the swamp, resulting in lower water levels and drying out of the peat. Development of free-flowing channels through the swamps (gullies) also prevents water from flowing evenly across the swamp surface, and sedimentation does not occur at the surface. Faster flowing channels increase the likelihood of erosion of the peat, further unbalancing the hydrology of the swamps (A Young 2010, pers. comm., October).

The fine-grained sediments of the swamps act as a water filter, controlling water quality and quantity to the swamp channels, and to down-gradient streams (Young & Young 1988). The larger swamps in the Woronora Plateau provide continuous flow to the main channels and to the streams emanating from the swamps (Tomkins & Humphreys 2006). This is in contrast to other streams in the area, which are typically ephemeral and only flow in response to rainfall (Tomkins & Humphreys 2006). Contributions to streamflow from smaller swamps are likely to be limited and seasonal (NSW DP 2008).

There are no monitored streamflow gauges to measure the volume of water released into the catchment from the peat swamps; however, from 1980 to 1988, discharge from the base of a headwater swamp in the Blue Mountains varied between 0 and 8.62 L/s (Holland et al. 1992). Studies of Zambian dambos (swamps) also indicate that the dambos contribute a significant portion of catchment surface run-off even though they cover only 5–10 per cent of the catchment area (Balek & Perry 1973). Since the swamps on the Woronora Plateau have deeper sediments and higher moisture content than the rest of the forested catchment, it can be assumed that they also contribute a significant portion of total catchment surface run-off (Young 1982).

Such changes to the swamp hydrology impacts the ecology of the swamps, with a transition to more terrestrial species frequently observed following swamp drainage. Drainage of sphagnum peatlands in southern Australia has resulted in the conversion of sphagnum bog to grasslands or sedgelands (Good 1992). Similarly, Nanson (2009) observed a community of peppermint gums becoming established on Drillhole Swamp on the Woronora Plateau as it dried out from gullying.

Under natural conditions, peat swamps are relatively protected from fire by the high water level, saturated nature of the peat and presence of wetland vegetation. Invasion of swamps by more terrestrial, woody species increases the available fuel for bushfires, potentially resulting in more intense fires (A Young 2010, pers. comm., October).

Headwater swamps are predominantly rainfall fed, and rely on rainfall exceeding evaporation to maintain saturated sediments within the swamp. Valley infill swamps are more likely to be fed by a combination of rainfall, stream flow and groundwater seepage, and, in some cases, discharge from the regional watertable (NSW PAC 2009). The dominant water source for hanging swamps is groundwater that seeps to the surface at cliff faces and steep valley sides.

3.4 Groundwater interaction

Information describing the natural groundwater regime of the peat swamps is limited, with few studies investigating the connection between the swamps and perched aquifers within the underlying sandstone. For example, in the Southern Coalfield, the watertable at the swamps is monitored in only a few locations (NSW DP 2008). There is significantly more discussion in the literature on the groundwater impacts to the swamps from subsidence than there is of the role of groundwater in the normal function of the swamps.

The sandstone formations underlying the swamps provide a low-permeability substrate, largely limiting the loss of water through the base of the swamp, although in places they may also feed shallow groundwater into the swamps through joints and bedding planes.

Groundwater movement in sandstone on the Woronora Plateau occurs mainly along bedding planes and fractures (Young 2007). Connectivity between groundwater and swamps therefore varies according to the presence of fractures and intersected bedding planes in specific locations (NSW DP 2008).

The flat, elevated topography at headwater swamps is more likely to result in swamps that are perched above the watertable (NSW DP 2008). Work at Drillhole Swamp for the Reynold's Inquiry (mid-1970s) measured groundwater 8 m below the bedrock surface of the swamp, indicating that the swamp was a perched system (Young 2007). Similarly, monitoring in the Kangaloon area by the Sydney Catchment Authority identified the watertable at 4–5 m below the swamps, which again indicates that the swamps were perched systems. These swamps rely on rainfall exceeding evaporation to maintain the perched watertable within the swamp sediments (NSW PAC 2009). Holland et al. (1992) and Young (1982) report that the water levels in swamps on the Woronora Plateau are higher than in the surrounding sandstone.

Valley infill swamps are located in more incised valleys and are more likely to be connected to groundwater within the horizontal sandstone units. Monitoring by Illawarra Coal of water levels within and surrounding Swamp 18 supports this hypothesis (NSW DP 2008). The water supply for these types of swamps is likely to be a combination of rainfall, stream flow and groundwater seepage, and, in some cases, discharge from the regional watertable (NSW PAC 2009).

Wingecarribee Swamp in the Southern Highlands is fed by groundwater discharge at the interface between the Hawkesbury Sandstone and an overlying basalt (Stricker & Stroinovsky 1995). These springs and stream flow from Caalang and Kangaloon creeks are the major source of water for the swamp (Coffey 2004). Butlers Swamp on the Southern Highlands is thought to receive groundwater discharge from the underlying sandstone aquifer; however, an associated aquifer test did not run for long enough to confirm the connection (URS 2007).

Hanging swamps in the Blue Mountains have the most obvious groundwater connection, since they exist in the sides of cliffs where groundwater emerges from the sandstone aquifer along a claystone layer. Although the groundwater seepage may not be continuous throughout the year, these ecosystems only exist because of the groundwater discharge and are therefore highly reliant on groundwater.

Overall, the reliance of the swamps on groundwater is variable. Some swamps are reliant only on rainfall or run-off trapped within the peat layers, some swamps are reliant predominantly on groundwater (such as the hanging swamps) and some swamps are reliant on a combination of surface water and groundwater.

3.5 Swamp water quality

Water within the peat swamps is largely contained in the peat; however, some free water also occurs in the narrow channels incised into the peat layers and, at certain times of year, occurs as shallow sheet flows or pools across the surface of the swamps.

Under natural conditions, the peat swamps are an acidic, reducing environment. Channel flow exiting from the swamps is also acidic. There is little information describing water quality within EPBC-listed peat swamps; however, some data are available for Martins Swamp on the Woronora Plateau and for stream channels exiting the peat swamps on the Woronora Plateau (Young 1982) and Barrington Tops (Mitsch & Gosselink 1986) (Table 3.1).

Table 3.1 Water quality data for two peat swamps.

Water quality parameter	Martins Swamp (Woronora Plateau)[a]	Martins Swamp pool[b]	Martins Swamp exit stream[c]	Barrington Tops exit stream[d]
pH	3.7–4.7	4.1–5.3	4.2–5.6	6.0–6.9
Dissolved oxygen (% saturation)	7–81	37–70	54–94	–
Organic carbon (ppm)	2.2–11.5	2.7–11.2	2.5–10.7	–
Silica (ppm)	0.7–4.5	1.0–4.1	1.4–4.3	–

– = not available; ppm = parts per million
a Data from Young 1982, water samples taken from swamp sediments.
b Data from Young 1982, water samples taken from a pool within the swamp.
c Data from Young 1982, water samples taken from an exit stream from Martins Swamp.
d Data from Mitsch and Gosselink 1986

Water quality within swamps is influenced by its level of organic and inorganic content, and its predominant water source.

Swamps that are located in cleared agricultural land may contain elevated levels of nutrients such as total phosphorus and nitrate/nitrite (SCA 2007). Waterways that drain into Wingecarribee Swamp are largely eutrophic. The peat of the swamp acts a sink for phosphorus, a major nutrient implicated in the development of algal blooms (AWT 1997). The swamp also appears to be an effective trap of suspended matter and bacteria carried by the inflows (AWT 1997). The Wingecarribee Swamp therefore acts as a water filter before water enters the Wingecarribee Reservoir. Salinity could occur in areas of poor groundwater quality, such as those associated with the Wianamatta Group (WSC 2011).

Nutrient enrichment has been identified as a key threat to the swamps in the Blue Mountains, particularly with the expansion and intensification of urban areas. The increased velocity, volume and nutrient content of urban run-off are likely to significantly increase rates of erosion, sedimentation and eutrophication, resulting in damage to swamp soils and native vegetation (NSW OEH 2011).

The literature review did not identify any water quality data specific to the headwaters upstream of the peat swamps. While the Sydney Catchment Authority monitors these catchments, monitoring is undertaken within the lakes and catchment streams downstream of the swamps. As such, the water quality upstream of these swamps has been inferred using the ANZECC and ARMCANZ (2000a) default trigger values for upland rivers. These rivers occur at altitudes above 150 m, and data collated to derive trigger values were from a number of sites, including eastern highland rivers that would exhibit similar characteristics as the headwaters of the peat swamps. Upland rivers generally have low nutrient concentrations and turbidity. Acidity levels are between pH 6.5 and pH 8, and although salinity of swamp headwater streams is dependent on catchment geology, it is generally low (55 µS/cm).

The swamps contribute to base flow and control downstream water quality of the streams they feed. Generally, plant matter decays and forms a dense organic mat on the swamp floor, which slowly releases water from the swamp to streams over an extended time (Toyer & Main 1981). Although data on the water quality of the swamps are limited, the quality of the downstream drainage lines is a good indicator. These show low levels of salinity (<1000 mg/L total dissolved salts) and a pH range of 4 to 8 (SCA 2006).

The majority of creeks in the Blue Mountains area exhibit consistently good water quality and favourable habitat conditions with good SIGNAL-SF scores (an index of water quality based on the presence of macroinvertebrates) and species richness (BMCC 2010, 2011). Sampling of the Wolgan River at Newnes and Wolgan Gap indicated that the river generally had high aquatic biodiversity and was in good condition. The Wollangambe River at its headwaters was in much poorer condition, possibly due to discharges from the coalmine. The quality of the river did, however, improve with distance downstream (NSW DECC 2008). Following rainfall, water quality is known to deteriorate from increased loads of nutrients originating from a variety of sources, including fertilised gardens, lawns and golf courses, industrial infrastructure, leachates and motor vehicles.

Historical water quality within streams of the Newnes Plateau is considered good, aside from pH and iron concentrations that occasionally exceed recommended levels. The streams are mildly acidic, with pH ranging between 5 and 6.3 (Toyer & Main 1981). Elevated iron levels have been observed, linked to a reddish precipitate on swamp floors formed through the cycle of iron reduction and oxidation by organic matter, bacteria and dissolved oxygen in the swamp water (Toyer & Main 1981). Young (1982) also recognised high iron content in water downstream from swamps and attributed it to in situ weathering of the sandstone bedrock.

Literature on swamp water in the Newnes Plateau characterises it as containing a high proportion of silica due to the sedge plants dominating the swamps, and having low levels of alkalinity. Low levels of alkalinity mean that swamp water has little buffering capacity, and any addition of acid (for example from mine water) may increase the acidity of swamp water, which is naturally weakly acidic.

3.6 Ecological characterisation

The THPSS endangered ecological community encompasses 1050 temporary or permanent swamp areas that occur on sandstone substrates at altitudes above 600 m (DSEWPaC 2012a). The community is naturally fragmented because of its occurrence in specific topographical locations. It occurs in two Interim Biogeographic Regionalisation of Australia bioregions: Sydney Basin and the South Eastern Highlands (DSEWPaC 2012a).

This highly variable ecological community may occur as several structural landscape forms, including hanging swamps, headwater swamps, valley infill swamps (defined in NSW DP 2008) and valley bottom swamps (mentioned in DSEWPaC 2012a). Vegetation associations in this community are highly variable due to its broad geographic range and the geological and hydrological gradients that influence its distribution. Broadly, the wetter parts of the swamps are dominated by sphagnum bogs and fens, while the drier areas are dominated by sedges and shrubs (DSEWPaC 2005). At least 19 threatened species protected under New South Wales and Commonwealth legislation are known to occur within the community, with 14 of these being plant species (Table 3.2) (DSEWPaC 2012a). The community also provides critical habitat for a range of endemic species, such as *Almaleea incurvata* and *Acacia ptychtoclada* (Carey 2007), although flora survey records show that these species are not present across all swamp types in the community.

The Blue Mountains water skink (*Eulamprus leuraensis*) is also considered to be endemic to this swamp community. Its presence is likely to be largely influenced by microhabitats formed as a result of localised topographical, geological and hydrological profiles. The skinks have low dispersal capability and, as such, are highly susceptible to any impacts on the swamps (Dubey & Shine 2010). Although information about vegetation associations across these landforms is lacking, some floristic assemblages have been previously described.

Table 3.2 Threatened species that occur within the Temperate Highland Peat Swamps on Sandstone endangered ecological community.

Common name	Scientific name	Conservation status	
		Cwlth EPBC Act	NSW TSC Act
Flora			
Bantam bush pea	*Pultenaea parrisiae*	Vulnerable	Vulnerable
Cord rush	*Baloskion longipes*	Vulnerable	Vulnerable
Deane’s boronia	*Boronia deanei*	Vulnerable	Vulnerable
Dwarf Kerrawang	*Rulingia prostrata*	Endangered	Endangered
Mountain swamp gum	*Eucalyptus aquatica*	Vulnerable	Vulnerable
Swamp bush pea	*Pultenaea glabra*	Vulnerable	Vulnerable
Tawny leek orchid	*Prasophyllum fuscum*	Vulnerable	Vulnerable
Wingecarribee gentian	*Gentiana wingecarribiensis*	Endangered	Endangered
Wingecarribee leek orchid	*Prasophyllum uroglossum*	Endangered	Endangered
–	*Carex klaphakei*	Not listed	Endangered
–	*Derwentia blakelyi*	Not listed	Vulnerable
–	*Eucalyptus copulans*	Not listed	Endangered
–	*Lepidosperma evansianum*	Not listed	Vulnerable
–	*Persoonia hindii*	Not listed	Endangered
Fauna			
Australasian bittern	*Botaurus poiciloptilus*	Not listed	Vulnerable
Blue Mountains water skink	*Eulamprus leuraensis*	Endangered	Endangered
Giant burrowing frog	*Heleioporus australiacus*	Vulnerable	Vulnerable
Giant dragonfly	*Petalura gigantea*	Not listed	Endangered
Red-crowned toadlet	*Pseudophryne australis*	Not listed	Vulnerable

– = none known, EPBC Act = Environment Protection and Biodiversity Conservation Act 1999, NSW TSC Act = New South Wales *Threatened Species Conservation Act 1995*
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Sphagnum-dominated peat swamps are rare in Australia because of the dry Australian climate (Whinam & Hope 2005). Where they do occur they are generally small and so are more sensitive to changes in hydrology, and have low species richness (Whinam & Hope 2005). Swamps in the Woronora Plateau rarely contain sphagnum, while in the Blue Mountains it is more common but still not a major component of the vegetation. Sphagnum is a major vegetation component at Wingecarribee Swamp (Figure 3.3), which is a relatively unique swamp in the context of the EPBC listing.

Figure 3.3 Sphagnum at Wingecarribee peat swamp, 2011 (widespread but not common in the Woronora area and Southern Highlands; restricted to permanently saturated places).

3.6.1 Blue Mountains and Newnes Plateau

3.6.1.1 Hanging swamps

Hanging swamps occur on steep valley sides and are formed by groundwater seeping between sandstone and claystone rock layers. This swamp type typically has low levels of sedimentation and accumulates organic material slowly, resulting in a shallow peat layer (DSEWPaC 2005). The hanging swamps of the Blue Mountains and Newnes Plateau support open heath vegetation communities dominated by shrubs and sedges. Common plant families include Ericaceae (subfamily Styphelioideae), Myrtaceae, Dilleniaceae and Restionaceae. Common species include coral heath (*Epacris microphylla*), blunt-leaf heath (*Epacris obtusifolia*) and pink swamp heath (*Sprengelia incarnata*) in the Ericaceae; pink tea-tree (*Leptospermum squarrosum*) in the Myrtaceae; guinea flower (*Hibbertia cistiflora*) in the Dilleniaceae; and spreading rope rush (*Empodisma minus*) in the Restionaceae (DSEWPaC 2012a). Regionally important plant species can occur in these swamps, such as bush peas (*Pultenaea glabra* and *Almaleea incurvata*), wattles (e.g. *Acacia ptychoclada*) and cut-leaved xanthosia (*Xanthosia dissecta*) (Environment Australia 2001).

3.6.1.2 Headwater swamps

Headwater swamps occur on gentle slopes and often occur across several benches or steps of the slope (Young 1982). Above the Metropolitan Colliery they support shrublands and heathlands, with sedges and rushes comprising the majority of the vegetation, with some occurrence of thickets of tea-tree and banksia. Riparian eucalypt woodland also fringes the swamps in some areas (HCPL 2008).

Common plant families that are represented in headwater swamps include Myrtaceae, Ericaceae (subfamily Styphelioideae), Restionaceae, Cyperaceae and some genera of Rutaceae and Proteaceae. Common species include baeckeas (*Baeckea linifolia* and *B. utilis*) in the Myrtaceae; swamp heath (*Epacris paludosa*) in the Ericaceae, spreading rope rush (*E. minus*) and razor sedge (*Lepidosperma limicola*) in the Cyperaceae. Deane's boronia (*Boronia deanei*), which is listed as threatened under New South Wales and Commonwealth legislation, is also found here. Scattered trees, such as Wolgan snow gum (*Eucalyptus gregsoniana*) or silver banksia (*Banksia marginata*), may also be present. Areas of permanent water or drainage lines support sedgeland vegetation (DSEWPaC 2012a).

3.6.1.3 Valley bottom swamps

Valley bottom swamps occur on valley floors in the main drainage line of the valley. They can be linear or branched, with many flow paths, discontinuous channel paths and pools, and may be perennial or ephemeral. Organic material gathers more quickly in this type of swamp, resulting in a deeper layer of peat. It is not clear from the literature whether valley infill swamps (as discussed in Tomkin & Humphreys [2006], NSW DP [2008] and NSW PAC [2009]) are different from valley bottom swamps (the terminology used in DSEWPaC 2012a, and largely used to inform this section).

Valley bottom swamps of the Blue Mountains and Newnes Plateau area generally support closed sedgeland communities with occasional shrubs. Plant families commonly present include Restioneaceae, Cyperaceae and Myrtaceae. Common species include spreading rope rush (*E. minus*), button grass (*Gymnoschoenus sphaerocephalus*) and razor sedge (*Lepidosperma limicola*) in the Cyperaceae, and woolly tea-tree (*Leptospermum lanigerum*) in the Myrtaceae (DSEWPaC 2012a).

3.6.2 Southern Highlands and Woronora Plateau

3.6.2.1 Hanging swamps

Hanging swamps also occur in the Southern Coalfield—for example, at Bargo and Cataract gorges on the Woronora Plateau (NSW DP 2008), which are below the minimum elevation limit of the EPBC community. The ecology of hanging swamps in areas outside of the Blue Mountains – Newnes Plateau area has not been previously described.

3.6.2.2 Headwater swamps

Headwater swamps (Figure 3.4) exist in the Southern Coalfield, particularly on the Illawarra and Woronora plateaus (NSW DP 2008). However, it is not clear whether any of the Southern Highlands swamps included in the EPBC listing are considered to be headwater swamps. Therefore, no information on the ecology specific to headwater swamps in the Southern Highlands is available.

Headwater swamps are the dominant swamp type on the Woronora Plateau (which lies outside of the THPSS community listing). The Metropolitan Colliery Environmental Assessment identified six vegetation associations, as classified by Keith and Myerscough (1993), associated with headwater swamps in the project area (NSW PAC 2009):

- fringing eucalypt woodland
- banksia thicket

- restioid heath
- sedgeland
- cyperoid heath
- tea-tree thicket.

3.6.2.3 Other swamps in the Southern Highlands

Swamps in the Southern Highlands that are included in the EPBC-listed THPSS community include Wingecarribee, Butlers and Wildes Meadow swamps. These swamps have not been classified as either headwater, valley bottom or valley infill swamps. They generally support highly diverse mosaics of vegetation, including sphagnum (*Sphagnum cristatum*) mossland, and open and closed sedgelands, grasslands, heath, shrublands and tall shrublands. Plant families present include Cyperaceae, Juncaceae, Poaceae, Myrtaceae and Ericaceae (subfamily Styphelioideae). Within these families, species present commonly include tussock sedge (*Carex gaudichaudiana*) and spike sedges (*Eleocharis* spp.) in the Cyperaceae, rushes (*Juncus* spp.) in the Juncaceae, common reed (*Phragmites australis*) and tussock grass (*Poa* spp.) in the Poaceae, river bottlebrush (*Callistemon sieberi*) and woolly tea-tree (*Leptospermum grandifolium*) in the Myrtaceae and swamp heath (*Epacris paludosa*) in the Ericaceae (DSEWPaC 2012a).

Figure 3.4 Headwater swamp vegetation located at the head of Mount Hay Creek, Blue Mountains National Park (includes buttongrass (*Gymnoschoenus sphaerocephalus*), yellow-eyes (*Xyris ustulata*) and razor sedge (*Lepidosperma limicola*)).

The Wingcarribee Swamp is the largest peat swamp in New South Wales. It is surrounded on three sides by low basalt hills and contains a deep acid peat layer. The average depth of the peat in Wingcarribee Swamp is 3 m, but areas up to 10 m deep also occur (Kodela & Hope 1992). The dominant vegetation community is *Lepyrodia anarthria* open rushland. A wide variety of other vegetation communities are also supported, including open woodland, closed tussock grassland, tall shrubland, sphagnum mossland, closed sedgeland and open sedgeland. Notable species include the Commonwealth-listed Wingecarribee gentian (*Gentiana wingecarribiensis*), and a range of Commonwealth and New South Wales threatened fauna, including the giant dragonfly (*Petalura gigantea*) (endangered), Australasian bittern (*Botaurus poiciloptilus*) (vulnerable), green and golden bell frog (*Litoria aurea*) (endangered), koala (*Phascolarctos cinereus*) (vulnerable) and tiger quoll (*Dasyurus maculatus*) (endangered) (Environment Australia 2001).

Peer review comments on Chapter 3

Ann Young makes the following comments:

1. No standard classification of upland peat swamps exists, including for THPSS, and the relationship between the swamps as geomorphic entities (e.g. hanging swamps, headwater swamps) and as vegetation communities (e.g. shrub swamps, sedgelands) is not well documented. In addition, the THPSS endangered ecological community is not a group of uniform swamps. Dr Young has suggested an alternative geomorphic classification (Appendix B) for the Sydney Bioregion, with four types of upland swamps: hanging swamps, headwater swamps, valley floor swamps and valley infill swamps.
2. The deposition of sandy sediment in the upland swamps depends on low-gradient streams with inadequate capacity to transport all of the coarse sediment out of their subcatchments, and is enhanced when the dense vegetation growing on wet sediment disperses run-off across the valley floor. This is the antithesis of stable narrow channels, which is why the stable narrow incised channels described for Barrington Tops by Nanson (2006 and 2009) are so unusual.

4 Ecological sensitivity to longwall mining impacts

Ecological knowledge about Temperate Highland Peat Swamps on Sandstone (THPSS) is known to be incomplete, and inherent variability in THPSS exists across the geographic extent of the community (DSEWPaC 2012a). The sensitivity analysis (described further in Chapter 5) investigated the sensitivity of the ecological community as a whole and individual species within the community. This section reviews the current understanding of ecological response to subsidence impacts.

Individual species are likely to be more sensitive to specific changes in the peat swamp environment, whereas the community as a whole is expected to be more resilient. In other words, individual species could become extinct but, as a community, the peat swamps would continue to exist. The species for which sensitivity could be assessed are the giant burrowing frog (*Heleioporus australiacus*), Blue Mountains water skink (*Eulamprus leuraensis*), giant dragonfly (*Petalura gigantea*) and spreading rope rush (*Empodisma minus*) (see note 1 at the end of the chapter). These species were selected because information was available on their habitat requirements, which enabled the impacts of longwall mining on the critical habitat of the species to be assessed. Unless otherwise referenced, information about these species has been taken from the New South Wales (NSW) Office of Environment Threatened Species Profiles.[2]

This section also discusses the potential impacts of coalmine waste water discharge on the swamp ecology (Section 4.2).

4.1 Sensitivity to subsidence impacts

4.1.1 The ecological community as a whole

The conceptual model for the THPSS ecological community as a whole included consideration of the interactions of the flora, fauna and other living organisms that coexist under the biophysical conditions that create that particular environment. Data that specifically describes the overall ecological response to change in the peat swamp environment is lacking, and the inherent variability of those swamp environments (and the microhabitats within them) make it difficult to model the community as a whole.

However, there are a range of common plant families that occur across all of the swamps described in the EPBC listing—for example, Restionaceae, Cyperaceae, Myrtaceae and Ericaceae (subfamily Styphelioideae)—and they may be used to indicate sensitivity of the community to change. The community's vegetation types may be arbitrarily categorised into three structural types:

- most susceptible to subsidence impact, including lower growing vegetation types (e.g. aquatic vegetation, sphagnum bogs, fens) with shallower root networks and a higher substrate moisture dependency

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[2] www.environment.nsw.gov.au/threatenedspeciesapp

- moderately susceptible to subsidence impact, including medium-growing sedge/herb/shrub-dominated vegetation types with moderately deep root networks and smaller substrate moisture dependency
- least susceptible to subsidence impact, including higher growing rush/shrub/tree-dominated vegetation types with relatively deep root networks and even smaller substrate moisture dependency.

There will, however, be exceptions to this general categorisation. No evidence has been identified in the literature that reinforces the suggestion that the depth of the root network and dependence of vegetation on substrate moisture influence the vegetation's susceptibility to subsidence (see note 2 at end of the chapter). However, further investigation of literature or field testing are needed to establish a link (G Sainty 2012, pers. comm., 13 November). Detecting vegetation change due to the effects of subsidence may be confounded by the time lag that can occur between the time of impact to the detectable response exhibited by the community's vegetation (M Krogh 2012, pers. comm., 24 October).

4.1.2 Fauna

Three threatened fauna species have been included in the conceptual model for temperate peat swamps. The threatened status of these species has arisen because of their specific habitat requirements. Changes to the peat swamp systems are considered likely to have the largest effect on these species. Unless otherwise referenced, information about these species has been taken from the NSW Office of Environment Threatened Species Profiles.[3]

4.1.2.1 Giant burrowing frog (*Heleioporus australiacus*)

Listed as vulnerable under both the New South Wales *Threatened Species Conservation Act 1995* (TSC Act) and the Commonwealth EPBC Act, the giant burrowing frog (Figure 4.1) is a large, rotund, slow-moving frog that is known from two distinct populations in NSW and Victoria. In NSW, the population is confined to the sandstone environments of the Sydney Basin, extending south to Ulladulla.

As its name suggests, the frog burrows below the soil surface or in the leaf litter across a range of non-breeding terrestrial habitats, including heaths, woodlands and dry sclerophyll forest on a variety of soil types, except those that are clay based. Breeding tends to occur in autumn, although calls have been recorded year round. Egg masses are laid under vegetation or rocks, or in burrows at soaks or pools in first- or second-order streams are expected to be occupied for up to 10 days immediately before or after heavy rain. Subsequent rainfall washes tadpoles into larger pools. Consequently, surface water flows and ponding are critical to the viability of the species. The frog has a generalist diet comprising invertebrates, so maintaining site ecological integrity to support naturally occurring invertebrate populations during breeding is important to ensure the survival of the species. Based on the known ecology and biology of the giant burrowing frog, general loss of breeding habitat through altered hydrological regimes and fire are its greatest threat.

[3] www.environment.nsw.gov.au/threatenedspecies

Figure 4.1 Giant burrowing frog (*Heleioporus australiacus*).

4.1.2.2 Blue Mountains water skink (*Eulamprus leuraensis*)

Little is known about the biology and ecology of the endangered Blue Mountains water skink (Figure 4.2); however, the semi-aquatic species is known from fewer than 40 locations between Newnes Plateau and Hazelbrook within two genetically distinct populations (Newnes Plateau population and Blue Mountains population). Local populations have also been found to be genetically distinct, even between populations less than 500 m apart. Dispersal has been rarely observed, suggesting recolonisation after disturbance is likely to be low or non-existent. Females give birth in early summer to live young. The habitat of the species is highly restricted and confined to isolated and naturally fragmented sedge and shrub swamps that have boggy soils and appear to be permanently wet. The vegetation in these swamps is typically sedgeland interspersed with shrubs, but may be a dense shrub thicket. Tussock grasses and holes are expected to provide shelter from predation. The water skink's diet appears to be dominated by invertebrates (grasshoppers, flies, moths, weevils and wasps), so maintaining site ecological integrity to support naturally occurring invertebrate populations is important to ensure the survival of the species. Given the very restricted distribution of the species, any impacts to swamps could have irreversible impacts to the skink. Critical risks would include lowering of the watertable and any changes in water quality that might impact on food resources. Loss of vegetation in individual swamps due to subsidence is also likely to reduce the amount of habitat available to the species.

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Figure 4.2 Blue Mountains water skink (*Eulamprus leuraensis*).

4.1.2.3 Giant dragonfly (*Petalura gigantea*)

The giant dragonfly inhabits permanent swamps and bogs along the east coast of NSW that have some free water and open vegetation. It is absent west of the Great Dividing Range. The species has been observed in swamps of the Blue Mountains, Southern Highlands and Clarence River catchment, and in some coastal swamps. Its ecology and biology suggests that the dragonfly's entire lifecycle is intrinsically linked to peat swamp environments. Adults are poor flyers, roost in vegetation on swamp margins and fly over open water to forage. Their diet largely comprises invertebrates hunted from the swamp. Males have been observed to congregate in vegetation waiting for females to mate with. Groundwater seepage areas in the swamp provide soft and moist microhabitats (including moss, under other soft ground layer vegetation or into moist litter and humic soils) for females to lay eggs into. Once hatched, larvae of the giant dragonfly dig long, branching burrows under the swamp. The larvae are semi-terrestrial, living in burrows in the peat for between 10 and 30 years. They leave their burrows at night to feed on insects and other invertebrates on the surface, and use underwater entrances to hunt for food in the aquatic vegetation. Almost any changes to the integrity of peat swamps in the region could conceivably have an impact on the viability of the giant dragonfly. Most significant would be any changes to peat stability that could affect nymph burrows, and alterations to watertable levels that could expose or drown burrows. Any changes to the health and composition of vegetation within the swamp may also affect foraging opportunities for mature dragonflies.

4.1.3 Flora

Information on the specific habitat requirements of vegetation species is limited, but one species could be modelled in a Bayesian belief network.

4.1.3.1 Spreading rope rush (*Empodisma minus*)

Spreading rope rush (*E. minus*) was chosen for inclusion in the model for sensitivity because the literature indicates that this species is reasonably expected to occur across all structural landscape forms associated with the THPSS endangered ecological community. The habitat requirements and ecology of this species are well documented and so provide a level of confidence associated with the model outputs.

Rope rush is a member of the Restionaceae family, comprising rush-like flowering plants native to the Southern Hemisphere. Plants in this family typically grow in nutrient deficient, moist environments, including peatlands, bogs, fens, wet heathlands and possibly stream banks. Species in this family are resilient to some change and exhibit adaptability to the climatic extremes of flood and drought (Linder & Rudall 2005).

Because of its resilience and adaptability, spreading rope rush is expected to occur across all swamp types within the THPSS endangered ecological community. It is a mid-to-late successional wetland species and a major peat former (except in areas where sphagnum moss dominates). Plants have a rhizome, with a cluster of roots that form a thick surface layer of about 50 mm (Clarkson et al. 2009) capable of retaining water up to 15 times their dry weight (Wagstaff & Clarkson 2012). As a result, this species is valuable to swamp ecosystems because it maintains soil stability and supports biomass for other macroflora (Wagstaff & Clarkson 2012). Rope rush is thought to obtain its nutrients from rainfall and atmospheric particulates by preferentially accessing the primary nitrogen input from rainfall with their cluster of roots (Clarkson et al. 2009). It is reasonably resilient to fire, resprouting from rootstock.

The biological and ecological characteristics of the species suggests that the greatest threats to its viability are nutrient enrichment as a result of changes to hydrological regimes, which will lead to increased competition from invasive species. This could be reasonably expected to occur in areas of poor water quality where pollutant concentrations may increase as a result of changes to inundation levels. Magnitudes of change of fire regimes (increase or decrease), may also impact on the species. Climate change and trampling or browsing by feral animals are also considered threats to the species.

4.2 Sensitivity to mine waste water discharge

Although the sensitivity analysis described in Chapter 5 is focused on the impacts of subsidence on the peat swamps, another recognised mining-related impact is waste water discharge into drainage lines uphill from the swamps. This section discusses the potential impacts on peat swamp ecology from releases of mine waste water above the peat swamps. There is little information available on either mine water quality or swamp water quality and, as such, it is difficult to be categorical about water quality and identification of potential impacts.

Mine waste water was previously discharged above ground into swamps at the headwaters of the Wolgan River. This, and probable impacts from rock fracturing and groundwater loss, resulted in significant impacts to ecology in East Wolgan, Narrow and Junction swamps (Springvale Coal & Centennial Angus Place 2011) (Figure 4.3). Discharges continued at least until 2009 in the Newnes area, but it is considered unlikely that it would be allowed to occur again. The waste water discharge had obvious impacts on the health of swamp vegetation; however, studies into the impacts were inconclusive and did not specifically link water quality to specific vegetation responses (M Krogh 2013, pers. comm., 3 December).

Figure 4.3 Impacts on East Wolgan Swamp from mine water discharge.

4.2.1 Mine waste water quality

Mining can impact on peat swamp water quality and ecology in both the short and long term. Typical impacts include:

- elevated turbidity and sedimentation from run-off due to site clearing and construction operations (generally short term)
- chemical pollution as a result of mine waste water discharge or coal stockpiles and refuse (long term)

- acid mine discharge due to the sulfur composition of the coal (long term). Release of acidic waste water can lead to low pH; high concentrations of sulfates, calcium, magnesium, iron and manganese; and minor amounts of trace elements in streams
- increased trace elements, including heavy metals, that may be released from coal deposits (generally long term).

Release of high-salinity waste water and acid mine drainage are high-priority issues with regard to waste water discharge from coalmines. The coalmines in the Hunter Valley discharge waste water to the Hunter River that must be kept below 900 µS/cm (Day & Riley 2004).

The chemical composition of mine water differs between mines. Generally, mine water is highly saline and can be discharged at various volumes. A case study of a longwall mine that had an average mine water discharge rate of 3.06 ML/day, showed a total dissolved solids concentration ranging between 5730 and 6810 ppm (Firth et al. 2002).

Mine water typically has higher concentrations of iron, manganese, potassium and zinc than occur in natural waterways. Potassium in mine water can also infer the presence of soluble longwall fluid and other petroleum products used in the underground coal extraction process (ACARP 2000). Overall, the quality of mine water may be influenced by its retention time in underground workings.

4.2.2 Potential impacts on ecology

Although there is literature on the impacts of mine water discharge on aquatic ecology (e.g. Rai 2009; Staniszewski & Jusik 2013), there is limited information available on the impact of mine discharge on peat swamps.

A number of water quality parameters that can be toxic to aquatic organisms, and can be measured by ecotoxicity testing include:

- pH: metal speciation that is deleterious to organisms
- bicarbonate: this ion contributes to salinity, and high concentrations are thought to be toxic to aquatic life, although the range at which it becomes toxic is narrow. Long-term exposure to low concentrations can also impact on ecology—studies have recorded histopathological lesions and diseased organs of both fish and invertebrates exposed to low bicarbonate concentrations (NSW OEH 2012a)
- trace metals: the effect of trace metals on ecology is dependent on the type and concentration. Concentrations of metals will vary depending on the instream dilution effects at the time of mine water discharge. The type of water (soft or hard) can also influence metal toxicity. For example, zinc and nickel can be toxic in soft water; however, their toxicity, particularly of zinc, is reduced in hard (high bicarbonate concentrations) water (NSW OEH 2012a)
- oil: although only toxic in high concentrations (>10 mg/L), soluble oil can affect aquatic organisms through biodegradation. As oil biodegrades it uses up oxygen, and oxygen depletion can affect (and often lead to the death) of aquatic organisms (ACARP 2000)
- salinity: similar to metals, saline water discharged from mines generally decreases in concentration with distance downstream. Salinity can affect organisms directly through osmotic stress, or indirectly through changes in habitat and food resources. It is a key stressor to aquatic freshwater organisms that live in freshwater environments. Studies have shown that the effects of salinity on biota may be reduced, depending on the variety of ionic constituents present in the mine water (Cardno 2010).

Coalmine discharge can alter the spatial and temporal variability in the distribution and composition of aquatic organisms. Overall, the effects of discharge decrease with distance downstream (although this is dependent on the extent of instream dilution). This is supported by studies that have shown an increase in pollution-sensitive taxa with increasing distance from the point of discharge (Cardno 2010). In terms of impacts on different organisms, generally, macroinvertebrates are more tolerant, particularly to increased salinities than other aquatic organisms, such as diatoms.

Only a limited amount of the information available on the response of macrophytes to mine water discharge is applicable to peat swamps in NSW. Generally, it is thought that mine discharge can pose a risk to vegetation because of its high salinity and elevated iron concentrations. Cardno (2010) reported that the organic mass of the macrophyte *Triglochin procerum* decreased as electrical conductivity increased. The rate of decrease is dependent on the presence of bacteria that can convert lignin into organic compounds (Cardno 2010). Mining can also result in iron precipitates and iron-oxidising bacteria that can lead to loss of native plants from toxicity or smothering (NSW DECC 2007; see note 3 at the end of the chapter).

Derivation of water quality trigger values for toxicants is based on ‘no effect’ concentrations determined from multispecies ecotoxicological assessments. Assessment of the effects of mine water discharge needs to consider a number of variables related to local geological, hydrological and environmental conditions and to the different chemical compositions of mine discharge water (Cardno 2010). As well, the responses of biota to mine water can differ between seasons and stream types (ephemeral or permanent). The unique nature of environmental conditions in different places precludes the direct application of results from studies that determine site-specific trigger values (Cardno 2010). Where insufficient information is available to derive site-specific water quality trigger values that reflect local conditions, the ANZECC and ARMCANZ (2000a) guidelines provide the best approach to managing environmental impacts of mine water.

An example comparison of water quality stressors is shown in Table 4.1 for Gujarat NRE’s Russell Vale site. The mine has three licensed water discharge sites that release water to Bellambi Creek, only one of which has water quality restrictions specified in the discharge licence. These limits and the corresponding ANZECC guideline values (ANZECC & ARMCANZ 2000a) are shown in Table 4.1. Note that the waste stream is a combination of stormwater and mine water. The waste water discharge operates within the specific licence restrictions. Sampling in Bellambi Creek shows that generic water quality objectives for aquatic ecosystems (from ANZECC & ARMCANZ 2000a) are largely maintained for pH and total dissolved solids, but are exceeded for total Kjeldahl nitrogen and total phosphorus. These elevated results may reflect natural variation in the water quality of Bellambi Creek, or may be due to impacts of waste water discharge. Metals concentrations are also measured in Bellambi Creek and compared to ANZECC and ARMCANZ (2000a) guideline values for aquatic ecosystems, but concentrations in the waste water discharge stream are not reported.

Although this comparison provides a useful reference of generic water quality trigger levels for aquatic ecosystems, it does not provide information on the ecological tolerance of peat swamp ecology to mine waste water releases. The approach taken at Dendrobium in the Southern Highlands sets out water quality trigger values based on the predicted level of impact (BHP Billiton 2012), rather than known toxicity of contaminants to peat swamp ecology.

Table 4.1 Limits of waste water discharge from Russell Vale coal mine to Bellambi Creek.

Pollutant	100th percentile concentration limit[a]	Russell Vale discharge point[a]	ANZECC guideline values[b]	Bellambi Gully[a]
pH	6.5–9.2	7.1–9	6.5–8(9)	8.1–9.2
Oil and grease (mg/L)	10	<0.1	NS	<0.1
Total dissolved solids (mg/L)	NS	1111–1900	125–2200	1220–1900
Total Kjeldahl nitrogen (mg/L)	NS	0.4–1.1	0.5	0.4–0.9
Total phosphorus (mg/L)	NS	0.03–0.12	0.05	0.08–0.3
Total suspended solids (mg/L)	50	13–27	NS	1–52

NS = not specified
a Data from Gujarat NRE Coking Coal Limited 2012
b Data from ANZECC and ARMCANZ 2000a

Peer review comments on Chapter 4

1. Ann Young comments on the use of *Empodisma minus* in the sensitivity analysis:

   I am aware that it is a major peat-forming species in New Zealand and that it is widespread through the THPSS and also the Coastal Upland Swamps. However, no data is presented on how important it is as a component of the THPSS, or how variable this importance is between the Blue Mountains and the Southern Highlands swamps, and thus how good an indicator of change it might be. There is no data about which is the most important sedge to conserve in order to protect the integrity of the swamp ecology. By contrast, species of ferns have been identified recently as rapidly indicating impacts of dehydration due to mining-induced subsidence. These have been coral fern *Gleichenia microphylla* and king fern *Todea barbara.*

   In response, the authors comment:

   E. minus was selected as being suitable for use as an indicator species in the ecosystem sensitivity analysis as, at the time of writing, there was available supporting information on the presence of *E. minus* in each of the swamp types, including literature (e.g. Commonwealth resources specific to this community) and accumulated field experience in these environments. This species is tolerant of some environmental fluctuation. The suitability of using the suggested ferns as indicator species has been discussed with a representative from the Royal Botanic Gardens, Sydney. It is suggested that these ferns may have a lesser likelihood of occurrence at all swamps protected under the EPBC Act listing and be more responsive to climatic variations than *E. minus* rather than subsidence imposed impacts. Hence, the interpretation of environmental responses of some ferns may be more complicated than in *E. minus*. Further research is required to establish the importance of *E. minus* to the function of the THPSS and how variable this importance is between the various swamps included within the THPSS community listing.

2. *Ann Young comments on the categorisation of vegetation types most susceptible to subsidence impacts:*

   Some of the tallest shrubs in upland swamps are in the tea-tree thickets that grow along the very wet valley axes and thus would be very susceptible to a drop in watertable. The vegetation of hanging swamps on cliffs is only low-growing but highly water dependent. Vegetation height is <u>not</u> a surrogate for susceptibility to change in watertable. Nor do we know much at all about rooting depths of the flora.

Derek Eamus, comments:

These three vegetation types were distinguished on the basis of root depths. Within the literature on groundwater-dependent ecosystems there is a literature that examines root depth and access to groundwater and sensitivity to fluctuations in groundwater (i.e. the saturated zone, as seen, for example, in swamps). Thus Canham et al. (2012) examine phreatophyte root growth relative to fluctuating watertables in WA, while Naumburg (2005) investigated the impact of groundwater depth fluctuations in ecosystem response modelling and discuss root depth as a differential variable. Similarly the special edition of the Australian Journal of Botany (2006, volume 54, issue 2) contains several papers that support the idea that rooting depth and sensitivity to declining availability of groundwater might be related.

3. *Ann Young comments in relation to discharge of contaminated mine water*:

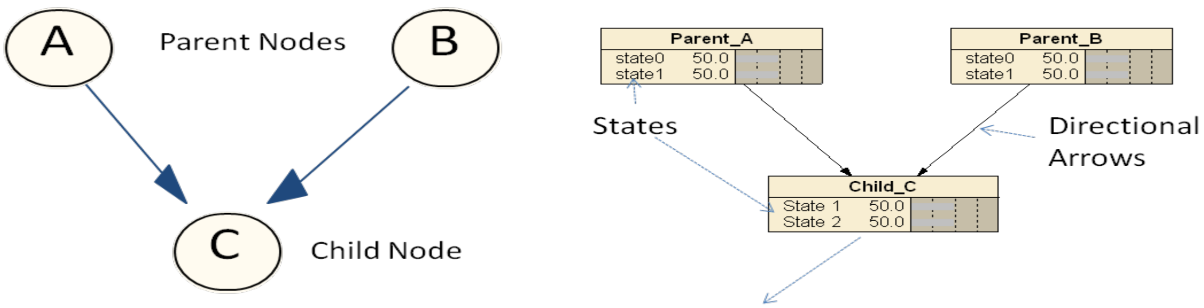
The more subtle impacts are potential changes in water quality parameters due to changes in stream discharge or to changes related to subsidence. For example, while iron concentrations can be high the precipitation of iron oxides and flourishing of iron-oxidising bacteria and resultant smothering and toxicity (reference NSW DECC 2007) due to mining is not from mine water discharge but from release of iron from shattered rock and from high groundwater flow. This is well-attested at Waratah Rivulet and many other undermined swamps.

5 Peat swamp ecological sensitivity analysis

To gauge ecological sensitivity of peat swamps, a Bayesian belief network (BBN) modelling was chosen. BBNs are a graphical technique used to aid decision-making and are suited to situations where limited information is available.

5.1 Introduction to Bayesian belief network modelling

BBNs were used to model the impacts of longwall coalmining on Temperate Highland Peat Swamps on Sandstone (THPSS) communities. In ecological applications, a BBN is an influence diagram that depicts logical or causal relationships of different environmental or management factors that can influence the likelihood of an outcome (Marcot et al. 2001). The influence diagram is made up of a graph with a set of connected parameters (henceforth called nodes), where directed connections from parent nodes to a child node indicate the parent node is having a causal influence on the child node. Each node is split into a set of categories (henceforth called states) to represent all values of that node. A BBN uses Bayes theorem and the chain rule from probability theory to update child nodes when the probabilities of a state for the parent node is specified (Stewart-Koster et al. 2010). The BBN uses conditional probability distributions to define relationships between the variables and the states within the variables (Ames et al. 2005). Probabilities are developed for each combination of states for each node that is linked to an outcome (or child node). Figure 5.1 clarifies the terminology used to discuss the components of a BBN.



Parent A	Parent B	State 1	State 2
State0	State0		
State0	State1		
State1	State0		
State1	State1		

Conditional probability Table (CTP)

Figure 5.1 Schematic representation of a Bayesian belief network, displaying the different components and terminology used in BBN analysis.

For risk management scenarios, BBNs are an effective tool for analysing and displaying likelihoods of certain outcomes, given the states of various factors. The ease of interpretation with BBNs allows the results to be communicated to a diverse audience, such as stakeholders and management bodies. BBNs have been effectively used in a range of fields to make qualified decisions based on sound knowledge (Marcot et al. 2001; Ames et al. 2005; Smith et al. 2007; Zhu & McBean 2007; Stewart-Koster et al. 2010).

Expert elicitation was used to model the complex relationships that describe the function of the peat swamps. Often field data with this information is insufficient or inadequate for the purpose, and resources are limited to supplement the data. An efficient source of less expensive information is knowledge gained from specialists with extensive experience. In the case of the peat swamps, there was no empirical data available and the BBN was therefore based entirely on the knowledge of specialists.

Specialists' knowledge has been increasingly incorporated into management recommendations and practices in a variety of fields (Yamada et al. 2003; Martin et al. 2005, Seoane et al. 2005). It has been used successfully as surrogates to data when information is not available and as a priori information for developing models in many fields before data become available (Store & Kangas 2001; Yamada et al. 2003; Kuhnert et al. 2005; O'Leary et al. 2008, 2009; Murray et al. 2009). Expert elicitation frequently occurs through individual or group interviews where answers are later synthesised using various algorithms into one consensus probability distribution. For example, this may be achieved by aggregating answers from postal surveys or interviewing multiple experts gathered as a group, such as expert panels (Martin et al. 2005). These approaches fall into mathematical aggregation and behavioural aggregation categories, respectively (O'Hagan et al. 2006). Johnson and Gillingham (2004) suggest expert-based predictive models can be sensitive to variability in expert knowledge.

The background, development and scenario analysis for the BBN modelling has been reported comprehensively in the previous milestone report for this project *Temperate Highland Peat Swamps and Sandstone: ecological sensitivity to impacts from longwall mining* (SKM 2012). This section gives a summary of the model and results.

The lack of empirical evidence, coupled with limited information of specific ecological responses to subsidence limit the future use of BBNs. Namely, BBNs should be used as a risk assessment tool, rather than a definitive measurement of impact. It is important to recognise this limitation of the model, and to use the BBN results to:

- flag the risk of potential impact to community/species
- indicate areas for priority investigation.

The primary use of the model is to design appropriate investigations to confirm the sensitivities it suggests, and to inform monitoring approaches in areas likely to be undermined. The BBN provides a framework that can be updated in the future, as empirical evidence of impacts to peat swamps becomes available.

The BBNs indicated that the peat swamps were most sensitive to changes in inundation. In developing an ecological monitoring programme, the ecological response to changes in inundation should form the basis of the monitoring programme design. However, the lack of knowledge about ecological responses means that ecological monitoring should be combined with hydrological and subsidence monitoring, to enable early warning of potential impacts.

5.2 Model development

5.2.1 Development of baseline model structure

The baseline model structure is shown in Figure 5.2. The arrows indicate how changes in one factor influence the next factor.

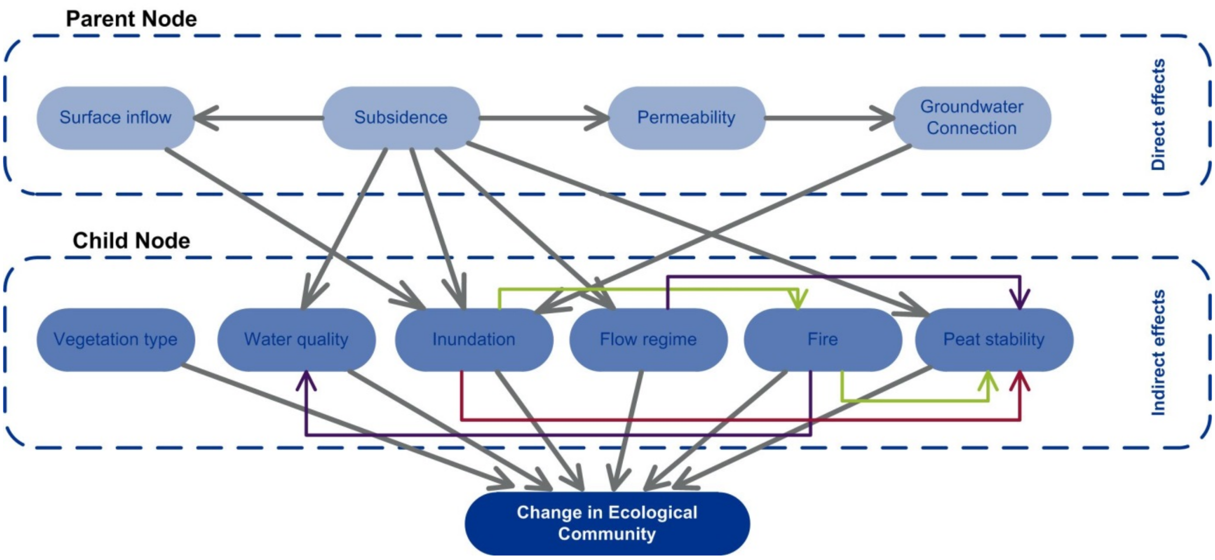


Figure 5.2 Baseline model structure for the Bayesian belief network capturing the direct and indirect effects of longwall mining on changes to the ecological community of peat swamps.

A change in the ecological community was affected by vegetation type, water quality, inundation, flow regime, fire and peat stability. Peat stability was in turn influenced by inundation, flow regime, fire and subsidence. Subsidence impacts were related to the geological characteristics of the area, mine dimensions, channel incision, mining depth and proximity of the mining activity. In turn, subsidence affected surface inflow and permeability of the rock as well as water quality, inundation, flow regime and peat stability.

The key understanding required to develop the model structure is how the various physical landscape components interact, and how longwall coalmining is likely to alter these interactions. As an example, groundwater flow to swamps occurs mainly through fractures, joints and bedding planes in the sandstone aquifers (Young 2007). Longwall mining can cause fracturing of the sandstone at the surface near the base of the peat swamp. The resulting increase in permeability alters the groundwater flow to the swamp, and is most likely to reduce groundwater discharge to the swamp. Similarly, swamps need a relatively stable level of inundation to avoid peat drying, shrinkage and erosion. Where subsidence occurs below a peat swamp, the increase in permeability can cause the swamp to drain, thereby causing the swamp to dry out and destabilising the peaty substrate. These are the types of relationships that are captured in the BBN.

Probability tables underlie the relationships in the BBN. The probabilities describe the likelihood of change in one factor influencing change in the next factor—for example, the likelihood that a decrease in groundwater connection will result in a change in inundation, or the likelihood that a decrease in inundation will result in impacts to the ecological community. Quantified data (i.e. from numerical modelling or field trials) was generally not available to inform the probabilities. However, useful qualitative descriptions were given in the literature which described the impacts to the physical environment caused by longwall mining. This qualitative data along with specialists' knowledge was sufficient to assign numbers (probabilities) to the relationships in the BBN.

All of the conceptual models (headwater swamps, valley infill swamps and hanging swamps) could be represented by the same baseline model structure confirmed in the specialist workshop. However, to distinguish the steeper topography and groundwater connection of hanging swamps and valley infill swamps from the flat topography and lack of groundwater connection for headwater swamps, a different set of probabilities was assigned for selected nodes. This effectively resulted in the development of two BBNs that described the three conceptual models:

- BBN1—for hanging swamps and valley infill swamps, with probabilities assigned to recognise the greater susceptibility to subsidence impacts engendered by the steeper topography and groundwater connection
- BBN2—for headwater swamps, where probabilities reflected the lower vulnerability of headwater swamps due to flat, elevated topography and lack of groundwater connection.

The resulting BBN modelling changes to the overall peat swamp community are shown in Appendix A along with the definitions for each influencing factor.

The scope required modelling of the sensitivity of the peat swamp community as a whole and modelling of individual species within the peat swamps. This required a basic understanding of how changes in the landscape caused by longwall mining can impact the ecology of the peat swamp. Data linking, for example, changes in inundation, peat stability and water quality to specific impacts on ecology is limited; however, habitat requirements for some species had been described in enough detail to be used in the BBN. These are described in Section 4.1 and are:

- THPSS ecological community
- giant burrowing frog
- Blue Mountains water skink
- giant dragonfly
- spreading rope rush.

The sensitivity of the species varied depending on their specific habitat requirements.

Information linking subsidence effects to ecological impacts is limited, with little information that specifically describes the ecological responses to changes in the surrounding environment. The lack of data necessitated a pragmatic and simplified approach to assigning probabilities to the ecological impact nodes. This approach was not used to assign probabilities to other nodes in the model because the interaction between other nodes is better understood (i.e. the interaction has been modelled, observed, reported) (see note 1 at the end of the chapter).

The simplified approach for identifying impacts to the community/species means the BBNs should be used as a risk assessment tool, rather than a definitive measurement of impact.

5.3 Sensitivity analysis results

Sensitivity analysis was used to determine the effect of each parameter on ecological community change and how 'sensitive' a model is to changes in model parameters. By measuring the uncertainty in the model, emphasis can be placed on parameters with enough sensitivity to affect the model results significantly when parameter values are changed. The sensitivity analysis captures the influence of the different parameters on the peat swamp community and the individual species, indicating which impacts the swamps are most sensitive to.

5.3.1 Ecological community model

In both the headwater swamp model and the hanging and valley infill swamps model, change to the ecological community was strongly influenced by the stability of the peat, and to a lesser extent by inundation and fire. This highlights that erosion of the peat has catastrophic impacts on the health of the peat swamp (see note 2 at the end of the chapter). The impact of fire is fully dependent on the level of inundation because the wetness of the swamp controls the frequency and intensity of fires that burn through the swamp. Therefore, the influence of these factors will always be similar.

In both models, peat stability was impacted by changes in inundation, to a lesser degree by fire and then subsidence impacts (i.e. cracking, tilting and/or fracturing of the underlying sandstone), and finally by flow regime. This reflects the conceptualisation of the peat swamps, which was that changes in inundation can have a significant impact on peat stability because it allows the peat to dry out, crack, fissure and erode.

Proximity has a profound effect on subsidence impact compared with the other factors affecting subsidence, since if a swamp is distant (i.e. more than 1 km) from the underlying mine workings, the likelihood of impact is significantly reduced. The other parameters that control the level of cracking, tilting and fracturing in the sandstone have a relatively equal influence.

The parameter ‘inundation’ was mainly influenced by the ‘change in groundwater connection’ for hanging and valley infill swamps, as these are more likely to be connected to groundwater than headwater swamps. The low (but not zero) sensitivity of headwater swamps to changes in groundwater connection reflects the possibility that some headwater swamps are fed by perched aquifers, but this is not the typical situation. ‘Subsidence impacts’ had a similar influence in both headwater swamps and hanging/valley infill swamps, which models the potential for the swamp to drain through subsidence-induced cracks and fissures in the underlying sandstone. Surface inflow has a lower influence on inundation for both swamp types because the swamps occur high in the catchment and subsidence-related impacts on the volume of surface water flows are likely to be much more significant further downstream from the swamps. The likelihood that subsidence-induced cracking and fracturing of the sandstone will reduce surface run-off to the swamps is limited in these high elevations and is less significant than the other influences on swamp inundation.

Water quality within the swamps is most sensitive to subsidence-induced cracking of the underlying sandstone, which could result in mineral dissolution, iron bacteria mats and methane gas release. Fire (which is more frequent and intense when inundation declines) also had a strong impact on water quality in both models, since fire burning through the swamp or through the surrounding catchment will significantly alter surface water run-off quality, which contributes to swamp water quality. Water quality susceptibility refers to the inherent water chemistry and whether it is likely to facilitate iron matting or floccing; however, this had less impact than subsidence impacts and fire.

5.3.2 Individual species models

Results for the modelling of the ecological community as a whole and the individual species are shown in Figure 5.3. The graph shows how sensitive a species is to each node. The higher the value, the more influence that factor has on the survival of that species.

Flora and fauna in all swamp types were most sensitive to changes in peat stability for both the community model and all the species models, since erosion of the peat destabilises the whole ecosystem. After peat stability, inundation has the strongest influence on community and faunal species sensitivity. Peat stability is also highly sensitive to inundation within the BBNs, since decreasing inundation dries out the swamp and causes erosion of the peat. Therefore, because inundation controls peat stability and fire (the other two strongest influences), it is the most important aspect of the swamp to maintain. Spreading rope rush is more influenced by water quality than are the community or faunal models. This result reflects the knowledge that spreading rope rush is affected by changes in water quality, such as nutrient enrichment, more than changes in fire or inundation.

The Blue Mountains water skink shows a more evenly distributed sensitivity to peat stability, inundation and fire than the other species modelled. This is most likely because the distribution of the species appears to be confined to permanently inundated swamp environments, with dispersal limited by drier environments. The other fauna species showed inundation to be only approximately half as influential as peat stability, although inundation also has a strong control on peat stability.

In the headwater swamps, peat stability is also the most influential parameter for all fauna models. Inundation was the next most important factor, followed by fire. Vegetation susceptibility and water quality impacts were the least important. Flow regime was also important to the frog. Vegetation susceptibility was again the most influential for the spreading rope rush; fire had the least effect.

The type of conceptual model made no difference to the relative level of influence of the parameters, with headwater and hanging/valley infill swamps being the most sensitive to the same parent nodes.

In summary, peat swamps generally have the highest sensitivity to changes in peat stability. However, peat stability is strongly influenced by inundation, since a decrease in inundation can cause drying, cracking and erosion of the peat. Maintaining inundation is therefore critical to preventing impacts on the peat swamp. (See note 3 at the end of the chapter.)

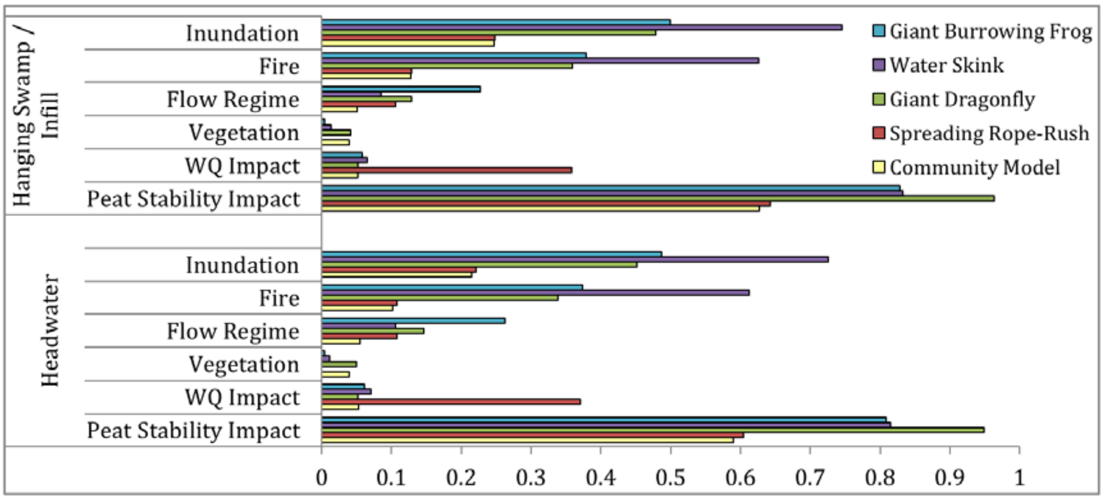


Figure 5.3 Results of the sensitivity analysis showing the entropy or variance reduction for the individual species models for both hanging/valley infill swamps and headwater swamps, when the ‘change in ecological community’ was adjusted for the probability of impact for individual species. The greater the value, the more influence the parameter had on the model.

5.4 Scenario analysis

The probabilities behind each influencing factor can be changed to model a high-impact scenario (i.e. when all factors are the most detrimental) and a low-impact scenario (i.e. when all factors are least detrimental to the swamps). This indicates whether the ecology can be preserved by managing mining parameters so that they are less detrimental.

The scenario analyses showed that the peat swamp ecological community and the individual modelled species were all impacted by the mining effects and associated ecosystem changes. The giant dragonfly appears to be the worst affected at high-impact scenarios but is also substantially affected with low-impact scenarios. The water skink and the burrowing frog are also greatly affected when impacts are high, but these levels can be reduced if the impacts are lowered. The peat swamp community had a similar effect. The rope rush is not as highly impacted as the other species.

Peer review comments on Chapter 5

1. *Both reviewers raised questions on the structure and linkages of the BBNs. For example, Derek Eamus has written:*
   I note that there was no link between groundwater connection and vegetation type, and no connection between surface inflow and fire. It would appear to me that such connections are likely to exist in the field and that therefore these should be incorporated into the BBN. *Ann Young has also questioned* the lack a link between groundwater connection and vegetation type, and between surface inflow and fire.

   Author's response:
   The process of identifying relevant variables and connections employs knowledge of the BBN model designers. In this case, expert elicitation was contributed by hydrology, ecology, hydrogeology and mining specialists convened for one day to workshop and develop the BBN model structure, identify and define variables, connections, their states, and evaluate subsequent probabilistic outcomes of the complex interactions that describe the function of the peat swamps. Limitations in applying BBN models exist and need to be acknowledged when interpreting outcomes. Outcomes generated by the BBN include those, and only those, explicitly described within the model structure. Other outcomes may exist beyond this model structure. Due to some of the acknowledged data gaps identified it was recommended that the primary use of the BBN models be to design appropriate investigations to confirm the sensitivities suggested and to inform monitoring approaches in swamp functions likely to be undermined by subsidence. The BBN provides a framework that can be updated in the future, as empirical evidence of impacts to peat swamps becomes available. As such, the modelled outcomes were recommended to be considered for applicability in future monitoring approaches along with further research and updated techniques. Regarding the Vegetation Type node it was defined as the states of vegetation susceptibility to change. As such, the connection, or not, of the groundwater node to the vegetation type node would have no bearing on whether the vegetation type would become more susceptible to change. However, the susceptibility of the vegetation type to change contributes to the potential for change of the overall community.

2. *Ann Young comments that* peat stability is not primarily affected by peat erosion. Peat dehydration and shrinkage are just as important and may have major ecological effects - loss of hydrophilic species, invasion of woody weeds, susceptibility to fires burning into the swamp sediments and peat, loss of habitat for stygofaunal species - before there is any erosion.

3. *Derek Eamus notes* there is a literature available that examines changes in inundation (frequency and depth) to swamp ecology**.** He suggests a review of the impacts of changes in swamp inundation (frequency and depth) would be highly relevant and informative to the overall aims of the report. If sensitivity of peat swamps to changes in water flow regime can be established, this could inform the determination of trigger values and also inform the design of monitoring. A number of swamp inundation references are listed in the References, under 'Additional references provided by peer reviewers'.

6 Evaluation of monitoring techniques

A range of methods are available for monitoring impacts on peat swamps and are appropriate for monitoring subsidence effects, changes to hydrology and impacts on ecology. This section briefly describes and evaluates the usefulness of each method for monitoring the impacts of longwall mining on peat swamps.

The focus of this report is on ecological monitoring to detect impacts; however, since geotechnical and hydrological methods are more important for early identification of impacts (while mitigation may still be possible), they are also discussed briefly here, with reference to more detailed studies.

Because peat swamps are often in steep and elevated terrain, many of the field-based methods are constrained by difficult site access, particularly if heavy equipment is necessary. Field methods are also constrained by the large areas that require monitoring and the number of swamps that exist within the area. For example, within the Bulli Seam Operations area, 226 swamps occur in an area of 220 km^2 (NSW PAC 2010). Therefore, fieldwork on individual swamps becomes resource intensive and time consuming. Remote-sensing methods offer benefits because they do not require site access and can more easily incorporate large areas; however, they can be limited by accuracy and resolution issues.

Because the advice in this document may be used by companies in developing monitoring programmes when they propose to longwall mine under swamps, this section is limited to discussion of monitoring methods that are in routine use (i.e. the equipment or data is available through commercial sources). Research techniques have not been considered.

The discussion recognises that monitoring will occur over a large area, for instance the mining lease. The evaluation of techniques must therefore consider the level of effort required to adequately monitor many discrete swamps over a large area, some of which may be inaccessible. The sensitivity analysis demonstrated that the most sensitive swamps are those near the edge of panels, in steeper topography and reliant on groundwater. As these swamps are the most vulnerable, they should be the focus of ongoing monitoring.

6.1 Methods for monitoring swamp ecological condition

A number of swamp attributes can be used to establish baseline ecological condition and can be monitored using a variety of methods. The methods outlined below—selected from the many available—are considered to be relevant to monitoring changes in swamp ecology. The limitation of using ecological variables as indicators of subsidence is that changes may also be attributable to factors other than subsidence, such as extreme weather events or fire. A detailed baseline of natural variability must be established for ecological indicators to be useful in detecting subsidence impacts. Another limitation is that ecological response to subsidence may be delayed by several years. It is therefore not useful as an early indication of impacts, or for setting trigger levels that can be used to implement management actions.

6.1.1 Vegetation survey methods

6.1.1.1 Flora species census

Plant species physiologically adapted to survive in periodically inundated conditions have a competitive advantage in wetlands and swamps. Alteration in swamp hydrology that causes drying of the peat will reduce this competitive advantage of wetland species and allow species assemblages to shift towards more terrestrial vegetation types. Increases in the proportion of terrestrial species in a swamp indicate changing swamp hydrology. No change in the proportion of terrestrial species (or change within equilibrium limits) indicates stability of hydrology and peat moisture levels.

Compiling a comprehensive species list will establish current condition, but change in vegetation composition occurs on a relatively long timescale (5–10 years), so this method is not suitable as an early warning of change. Furthermore, change in the proportion of wetland species can also be attributable to altered fire regime, climate variability or other disturbance.

Specific techniques for compiling a census of plant species are field based and require suitably experienced and qualified botanists to record species in plots, along transects or during random meanders for each impacted swamp. Random meanders involve randomly traversing the study area in a roughly back and forth pattern. In areas of preferred habitat for threatened species, the random meander technique allows for greater coverage than a plot-based survey and is less time consuming (NSW DEC 2004). Plot-based surveys consist of recording all species in a defined area. Transect surveys involve recording all species encountered over a defined linear distance.

The suitable number of plots/transects/meanders within each swamp will depend on individual swamp size and complexity. Species–area curves can be used to determine the number of transects/meanders or size of plots needed to obtain a comprehensive species list. A cumulative count of species is plotted against the number of plots/transects/meanders (or plot size, length of transect/meander). Beyond a certain number of plots/transects/meanders, the number of novel species identified only increases slightly. This is the point at which the curve flattens, and can be used to indicate when adequate survey effort has been made.

Survey plots/transects/meanders need to occur in all the subhabitats present to ensure the full range of species has been adequately sampled.

A plot-based survey is likely to be more accurate than transects or random meanders in swamp vegetation, because species are small, may be inconspicuous and detection requires a concentrated search over a smaller area. Also, a plot-based survey method is consistent with the threatened species and vegetation pattern surveys described below.

6.1.1.2 Monitoring vegetation community patterns

Vegetation communities within temperate peat swamps form a mosaic reflecting differences in inundation levels, frequency of wetting and the resulting soil/peat characteristics. Contraction or expansion of the area occupied by water-dependent species or vegetation will occur over time in response to changes in swamp hydrology.

Monitoring vegetation patterns is useful for establishing baseline conditions and variability, but not as an early warning indicator because of the time lag between hydrology changes and vegetation changes. Another limitation to using changes in vegetation patterns as an indicator of longwall mining impacts is that changes may be attributable to other factors (e.g. fire) unrelated to subsidence.

Methods for detecting change include field-based data collection and remote sensing. Seasonal data must be collected to establish what changes in vegetation patterns are attributable to seasonal influence. This is discussed further in the section on establishing natural variability (Section 6.4.5).

Determining vegetation patterns relies on identifying the different vegetation communities/associations within a study area and then mapping the distribution of each community. Data recorded to describe a vegetation community includes floristic and structural characteristics such as dominant species in each stratum, stratum heights and relative abundance of species. Relative species abundance is measured as stem counts, basal area counts, foliage projective cover (FPC), crown cover or Braun-Blanquet cover–abundance scores:

- stem counts: direct counts of the number of plants of each species in a plot
- basal area measurement: a plot-less method that involves using a Bitterlich gauge held 1 m from the eye while the observer turns 360 degrees and tallies plants with trunks that have a greater width than the gauge
- FPC: visual estimation of the percentage (of plot) occupied by the vertical projection of foliage
- crown cover: the percentage occupied by the vertical projection of the periphery of crowns, where crowns are treated as opaque
- Braun-Blanquet cover–abundance method (Braun-Blanquet 1965): involves choosing one of seven cover classes that correspond to a cover range. For example a score of 3 indicates the species covers between 25 per cent and 50 per cent of the surveyed area.

Direct stem counts are suitable for trees and large shrubs, but can be time consuming for sedges and herbs, and so are not suitable for peat swamps. The basal area method and crown cover estimates are better suited to woodland or forest vegetation types that to temperate peat swamp vegetation. FPC estimates are generally more accurate than assigning a Braun-Blanquet score, but more time consuming.

For peat swamp vegetation surveys, either the FPC or Braun-Blanquet method of recording species abundance is suitable, but because a large number of plots probably need to be surveyed, the Braun-Blanquet method is preferred because it is quicker.

Data can be collected from fixed-area plots or transects. Plots are more suitable for low-growing, densely packed vegetation such as sedgelands, rushlands and shrublands. The commonly used plot size in New South Wales is a 400 m^2 (20 m × 20 m); however, surveys of vegetation with smaller plants, greater plant density or greater species diversity require smaller plots (Sutherland 1996). The plot sizes in Table 6.1 are adapted from Sutherland (1996). To cater for the range of vegetation forms encountered in temperate peat swamps, a plot size of 25 m^2 is recommended.

Table 6.1 Recommended plot sizes for monitoring vegetation communities in peat swamps.

Plant form	Plot size (m^2)
Cryptogams	0.01–0.25
Grasses, herbs, short shrubs	0.25–16
Tall shrubs	25–100
Trees	400–2500

The boundaries of each vegetation type can be mapped using a global positioning system (GPS) unit in the field or by remote sensing. The accuracy of the GPS unit limits the scale of change that can be detected. For small swamps, differential GPS may be needed to detect changes at a meaningful scale, but the cost can be prohibitive. For larger study areas it can be more practical to use a combination of field data, aerial photography analysis, soil mapping and elevation data to map vegetation community boundaries, because the time needed to verify boundaries on the ground would be excessive.

Using remote sensing has the advantages of minimising field mobilisation resources and enabling use of historical data. Swamps that are difficult to access on the ground would be more suited to remote-sensing methods, which are discussed further in Section 6.2. Remote-sensing methods will need to be calibrated against field data, so some form of field programme will be necessary.

6.1.1.3 Monitoring vegetation condition

Desiccation and death of vegetation is associated with peat collapse and swamp dewatering. Lengthy periods of inundation also result in death of some plants. If the condition of swamp vegetation remains unchanged, this may indicate stable peat characteristics and unaltered water balance.

As with vegetation pattern changes, the limitation to using vegetation health as an indicator is that changes may be also attributable to factors such as extreme weather events, fire events and other disturbance unrelated to subsidence. Seasonal data must be collected to gain an understanding of what change in vegetation condition is within the expected natural range.

To establish a baseline, vegetation condition and peat condition for each swamp should be recorded. Methods of data collection can be field based or remotely sensed, or a combination of both. Field survey methods could be either plot-based surveys or transects. For plot-based surveys, the cover of dead vegetation, live vegetation and bare ground can be visually estimated as a percentage in fixed-area plots. Extent of fissures, cracks or peat oxidation may also be monitored using visual estimates. Visual estimation is generally more accurate in smaller (e.g. 1 m × 1 m) plots. A discrete point sampling method can be used in transects where live vegetation, dead vegetation or bare ground is recorded at 1 m intervals along a 100 m tape (Muir et al. 2011). Transects will not be suitable for all swamps because areas of inundation may make it impossible to traverse the swamp.

Remote-sensing methods using the normalised difference vegetation index (NDVI), which indicates vegetation vigour, and evapotranspiration data can also be used to monitor live and dead vegetation, extent of bare ground and peat cracks. Using remote sensing has the advantages of minimising cost and resources and enabling access to historical data. For swamps that are large, difficult to access or difficult to traverse, remote sensing is a more suitable method for monitoring vegetation condition; however, field verification would still be required. Remote sensing is discussed in Section 6.2.

6.1.2 Fauna monitoring methods

6.1.2.1 Wetland frog monitoring

Wetland frogs require ponding of water for breeding. Since the presence of wetland frogs at a swamp is dependent on continuing water availability, monitoring of wetland frogs is suitable for establishing baseline conditions at each swamp. As an indicator of change, loss of wetland frog species will lag behind swamp dewatering, especially for those species that only require water during the breeding season. Also, it is possible that loss of a species or group of species from a swamp could be attributed to other threatening processes, such as chytrid fungus, inbreeding in small populations, stochastic events (e.g. fire, drought) and pest animals. As such, disappearance of a wetland frog species from a swamp does not necessarily indicate subsidence impacts.

The threatened giant burrowing frog requires pools for breeding. The usefulness of this species as an indicator depends on the occurrence of a stable population, but as the species has only been found in the Blue Mountains swamps it is unlikely to be a useful indicator for all swamps that are part of the endangered Temperate Highland Peat Swamps on Sandstone Community.

Specific survey techniques for wetland frogs include visual encounter surveys, audio strip transects, night driving, pitfall traps, and visual larval and egg mass surveys. The methods are described in detail in *Survey guidelines for Australia's threatened frogs* (DEWHA 2010a). Suitability of each method for temperate peat swamps is evaluated in Table 6.2 (adapted from DEWHA 2010a).

Initially, species location records should be assessed to compile a list of likely species and their breeding behaviour. Species records can be sourced from other ecological surveys in the locality, and from the publicly available Protected Matters database (DSEWPaC 2011) and Atlas of NSW Wildlife (NSW OEH 2013).

To detect species, a combination of suitable methods (visual encounter surveys, audio strip transects , static call surveys and automated call recording) should be used during the breeding season, ideally during warm weather, after rainfall events and during periods of light wind. Repeated surveys are required to be confident that species present have been detected. For species such as the giant burrowing frog, where rates of occupancy per hectare are very low, the number of survey sites needs to be high to increase the probability of detection.

Table 6.2 Wetland frog survey methods.

Technique	Description	Target/ Species trait	Advantages	Disadvantages	Suitability for temperate peat swamps
Visual encounter surveys	Systematically searching a defined distance of suitable habitat for a prescribed time Queensland EPA (2005) recommends nocturnal and diurnal surveys with two observers searching 100 m × 50 m transects over 30 minutes Survey during the breeding season following heavy rain	Active or obvious species	Inexpensive, non-destructive, ideal for opportunistic surveys	Unsuitable for cryptic or secretive species	Not ideal for detecting giant burrowing frog, but generally suitable for other wetland frog species
Audio strip transects and static call surveys	Walking along designated transects (which traverse potential breeding habitats) with a tape recorder (manual recording)	Most species; especially good for prolonged breeders	Quick and non-destructive; may detect cryptic species	Only suitable during calling period, only detects calling males	Suitable for most wetland species but not for the giant burrowing frog, because it calls softly and

Technique	Description	Target/ Species trait	Advantages	Disadvantages	Suitability for temperate peat swamps
	to listen for calling male frogs				irregularly
Automated call recording	In the absence of an observer, uses a remote recording device, attached to a timer, to record calling frogs	Most species	Not labour intensive; recordings can be made over several days and in different conditions	Technical and equipment constraints; limited to area around recorder; equipment costs high	Suitable for most wetland frogs
Night driving	Slowly driving along quiet roads on nights when weather conditions are suitable for frog activity and recording frogs sitting on or moving across the road The car can be stopped for a defined period at intervals along the road to listen for calls	Large and small active species	Large areas can be surveyed in a short time; large and small species may be detected with visual and aural encounters	Requires road to bisect suitable habitat. Driving speed may affect the detection of smaller species	Not suitable as there is unlikely to be roads bisecting peat swamps
Pitfall trapping	Buried pipes or buckets are placed along a drift fence with variations in trap size, trap shape and drift fence length to accommodate variation in size of the target species or the type of habitat to be surveyed (DEWHA 2011). Queensland EPA (2005) suggests that leaving the trap for 10 days is most effective. Method determines presence/absence and does not estimate species abundance. Eyre et al. (2012) recommend a T-	Terrestrial and fossorial (burrowing) species	May detect cryptic species; can detect active but non-calling frogs	Not suitable for tree frogs (some tree frogs can be captured if funnel traps are added); expensive and labour-intensive installation; use may be limited by hard substrates; effectiveness dependent on weather conditions and season	Likely to be difficult to use due to underlying sandstone substrate. May be possible where peat has depth between 0.5 and 1 m Detection rates are very low when this method is used for the giant burrowing frog—approximately one animal in 800–1000 trap nights

Technique	Description	Target/ Species trait	Advantages	Disadvantages	Suitability for temperate peat swamps
	shaped design with four 20 L buckets and 7.5 m of drift fence between and beyond buckets. Open for four consecutive nights per survey period				
Egg mass surveys	Searching for egg masses in suitable microhabitats. A description of egg mass and deposition site for many species is provided in Anstis (2002)	Species with conspicuous eggs	Extends detection time for 'explosive' breeders; may detect cryptic species that are breeding but not calling	Not suitable for species with cryptic eggs; may have narrow temporal window for sampling	Not considered suitable in peat swamps as there is usually very little open water for eggs to be deposited in
Larval sampling	Sweeping a dip net through suspected aquatic microhabitats to capture and identify the species of larvae present at the site	Species with aquatic larval stage	Useful when adults are difficult to detect	Labour-intensive sampling of specific microhabitats; larvae are difficult to identify to species	Not considered suitable in peat swamps as there is usually very little open water for the larvae

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6.1.2.2 Reptile monitoring

Wetland reptiles are dependent on a water source at some part of their lifecycle. Species such as freshwater tortoises require a permanent water source and others such as skinks and snakes may only require a wetland habitat temporarily for foraging or breeding. Monitoring of wetland reptiles is suitable for establishing baseline conditions at each swamp; however, changes in presence/absence of a species may be attributable to impacts other than subsidence.

The threatened Blue Mountains water skink inhabits sedge and shrub swamps characterised by permanently wet sandy–peaty soil. Its presence can be used as an indicator of moisture availability in swamps. Field survey techniques include diurnal hand searches, visual searches and pitfall trapping, optimally undertaken in the summer months when the species is most active.

Diurnal hand searches can involve searching under rocks, fallen timber, leaf litter, bark, debris, or bark on the trunks of both living and dead trees. Since the Blue Mountains water skink shelters in dense tussocks and down holes, this method is not suitable because hand searches often cause destruction of habitat.

In a visual search, observing can be done from a distance with binoculars, while walking through suitable habitat, or while searching holes or cracks with a torch and/or endoscope camera. Dense groundcover in peat swamps is likely to make visual searches very difficult.

Pitfall trapping involves digging a PVC pipe or plastic bucket into the ground so that the lip is flush with the ground. Reptiles fall into the hole and cannot get out because of the smooth sides of the bucket/pipe. Traps can be checked daily or twice daily to record species. This method is recommended for the Blue Mountains water skink (DEWHA 2011) from December to February, when the species is most likely to be active. A line of three 10-L buckets should be spaced 5 m apart. This method may be difficult to implement in swamps where the peat is thin.

6.1.2.3 Wetland bird monitoring

Wetland birds require bodies of water for foraging habitat and aquatic vegetation (sedges/rushes) for shelter. Continuing presence of wetland birds can indicate the swamp hydrological regime is stable. Loss of wetland birds can indicate drying out of swamps. In particular, breeding activity declines in swamps that are drying out.

Survey methods suitable for wetland birds are described in detail in *Survey guidelines for Australia's threatened birds* (DEWHA 2010b). These include diurnal bird surveys, call broadcast surveys and nest surveys. Diurnal bird surveys are done in suitable foraging habitat in the early morning or early evening. All species sightings and calls are recorded in a fixed area, along a transect or from a point. A combination of transect and point methods is useful for the range of wetland bird species that could occur in peat swamps. Call broadcast surveys involve playing a recording of target species over a loudspeaker and detecting individuals of that species that respond to the call vocally, or are attracted by the call and observed as a result. Nest surveys involve counting nests in a fixed area during a species breeding season.

Diurnal surveys, call broadcast and nest counts are all suitable methods for peat swamps and should be used in conjunction to ensure the most comprehensive inventory of bird species is recorded. Repeated sampling over multiple days and at different times of the day improve detection and provide the best estimates of species richness at a site (Eyre et al. 2012). All surveys must be timed to coincide with the arrival or departure of migratory species (DEWHA 2010b).

Wetland bird monitoring is useful for establishing current condition and detecting change over time. However, a decrease in activity or visitation is unlikely to result immediately in response to a subtle change in hydrology, so is not suitable as an early warning indicator.

6.1.2.4 Aquatic macroinvertebrate diversity

Aquatic macroinvertebrate diversity can be used as a surrogate indicator for water quality (see Section 8.4.3). Sampling downstream from swamps can be used as an indicator to detect changes in water quality. However, there are many other variables that can affect macroinvertebrate diversity; therefore, it is difficult to draw conclusions from changes in macroinvertebrate diversity. In addition, many of the swamps are not associated with any open water and so the standard macroinvertebrate sampling techniques would not be possible.

Survey techniques consist of sweeping a standard-sized net through water, sampling riffle and edge habitats separately. Macroinvertebrates trapped in the net are then sorted to identify as many macroinvertebrate taxa as possible. Detailed methods are described in

Australia-wide assessment of river health: New South Wales AusRivAS sampling and processing manual (Turak & Waddell 2002). Environmental variables are also recorded at each sampling site. A software package containing predictive models is used to compare the sampling data with the expected diversity of macroinvertebrates.

The AusRivAS method is not suitable for monitoring peat swamps because there are no edge or riffle stream habitats to sample from. A more suitable method would be monitoring invertebrates. This method is discussed in the next section.

6.1.2.5 Wetland invertebrate monitoring

Wetlands and swamps provide habitat for insects that have an aquatic life stage—for example, dragonflies. Invertebrate species have different sensitivities to disturbance and hence can be used as an indicator of swamp condition and disturbance impacts.

Methods for invertebrate surveys include pitfall traps, beat sampling and hand foraging. Pitfall trapping involves digging holes in the ground and placing buckets in the hole with the rim flush with the natural ground level. Insects fall into the buckets and can then be identified. Hand-foraging techniques include sieving soil and leaf litter, raking through leaf litter and soil, searching in rock piles, and searching on trees trunks and under bark. Beat sampling involves spreading a sheet under vegetation and beating foliage with a stick so that insects drop out onto the sheet and can be identified. A detailed description of invertebrate sampling methods can be found in *Guidance statement 20: Sampling of short range endemic invertebrate fauna for environmental impact assessment in Western Australia* (WA EPA 2009). No one method is considered to be more effective or accurate than the others. Surveys should use a combination of methods to identify as many taxa as possible. Greater repetition of surveys will yield greater accuracy.

Targeted searches for the giant dragonfly should be included in invertebrate surveys. The combination of poor dispersal ability, long larval life and need for permanent swamp habitat with a stable watertable makes the giant dragonfly susceptible to alteration in swamp hydrology. Draining or flooding destroys the larvae in their burrows.

Specific survey techniques for the giant dragonfly include diurnal searches for adults and exuviae from late October to January using handheld sweep nets. The method determines presence/absence and does not estimate species abundance. Detection accuracy improves with number of successive visits. This species is only known in a few areas; hence suitability as a broad scale indicator is poor.

Monitoring of invertebrate species known to be particularly sensitive to water quality changes, such as some dragonflies and damselflies, may be suitable as an indicator of peat swamp condition if they occur across a broad area.

6.1.3 Invasive species

The appearance of invasive species or an increase in activity may be an indicator of altered swamp hydrology. Survey and monitoring invasive flora or fauna species can be used to establish current condition of threatening processes, as well as ongoing changes to swamp ecology. However, spread of invasive species is not useful as an early warning indicator due to the lag time between subsidence and the resulting impacts. In addition, the spread of weeds is also attributable to processes other than subsidence, such as land clearing, increased human visitation and fire.

Specific survey techniques for weed monitoring can be field based or remotely sensed. Field methods include measuring abundance of weeds in fixed plots and fixed-point photography.

Relative abundance of weed species can be measured as stem counts, basal area measurements, FPC, crown cover or Braun-Blanquet cover–abundance estimates. These measurements were defined previously in Section 6.1.1.2.

Fixed-point photography can be taken from a survey marker to visually record changes in weed density or abundance. Methodology involves ensuring the photo is consistently taken from the same point and elevation and in the same direction, and includes an easily recognisable object of known size such as a 1.8 m star picket, traffic cone or similar object. Photos are taken at regular intervals, such as biannually.

Data can be collected from fixed-area plots or transects. Plots are more suitable for low-growing, densely packed plant forms. The commonly used plot size in New South Wales is a 400 m^2 (20 m × 20 m) plot; however, surveys of vegetation with smaller plants, greater plant density or greater species diversity require smaller plots (Sutherland 1996). Plot size can be selected based on the life form of weed species that are being surveyed. Common weeds recorded in temperate peat swamps include trees, shrubs, climbers, grasses and herbs. As such, plot size will depend on the particular group of weeds present at each swamp. Nested plots can be used to ensure the various life forms are adequately surveyed, and that survey methodology is consistent. A nested-plot design is where smaller plots are located within the larger plot.

The extent of particular weed species may also be remotely sensed using NDVI if the spectral properties of the weed species can be differentiated from the surrounding native swamp vegetation.

Remote sensing is likely to be more suitable for large swamps, large weed infestations and swamps difficult to access. Field-based survey methods are most suitable for easily accessible small swamps with low-to-moderate weed infestation. Suitable field methods are a combination of Braun-Blanquet cover–abundance estimates and fixed-point photography, as this will enable rapid data collection.

Monitoring feral fauna presence/absence and activity is part of establishing the baseline for the effects of longwall mining; however, it is not directly correlated to subsidence impacts. Feral fauna survey techniques include baited camera traps, sand plots and visual assessment of damage (e.g. wallows). Baited infrared camera traps placed randomly in suitable habitat (at least one in a 100 m × 100 m site) are a useful tool to monitor the presence, abundance and activity of many feral animal species such as pigs, cats and foxes. Camera traps should be deployed for as long as possible, with a recommended minimum of four nights but ideally for longer than two weeks (Eyre et al. 2012). Sand plots involve laying a smooth damp sand pad across an unsealed road or forest floor (baited if desired to increase animal activity) and recording animal tracks. Sand plots are particularly useful for monitoring activity of feral animals (e.g. foxes and cats) that leave distinctive tracks. Activity indices (e.g. the Allen activity index) can be calculated from the number of tracks; however, these are not always good measures of animal abundance. Visual assessment of damage by feral animals can be made by estimating the extent of damage in permanent or fixed plots.

All three methods can be used to determine presence/absence but not species abundance. Sand plots are less likely to be useful for monitoring in peat swamps as there is unlikely to be a suitable open ground on which to place the smoothed sand. A combination of camera traps and damage estimates are most suitable because this will monitor the species that are causing the damage and the extent of impact they are having.

Monitoring invasive species will help to categorise the swamps in terms of condition and threats; however, this variable is not suitable as an early warning of impacts.

6.2 Remote sensing to monitor ecological response

Field surveys are limited to monitoring at specific locations and rely on data extrapolation to infer information outside monitoring areas. Airborne and satellite-based remote-sensing techniques offer alternative techniques for ecological monitoring when used in conjunction with ground-based methods, and lend themselves well to consistent, repeatable methods of analysis.

Remote-sensing methods offer the advantages of continuous datasets over broad areas and the availability of repeat surveys facilitating historical and temporal monitoring within the mining zone and the greater surrounds. Techniques relevant to monitoring ecologic response include:

- passive (optical sensing) remote sensing through multispectral digital sensors, which can be used to investigate changes in vegetation community pattern, vegetation condition, boundary extent and erosion
- active (laser/radar emitting) remote sensing through lidar and radar, which can be used for vegetation condition and boundary extent.

These techniques are evaluated in this section.

One of the benefits of remote sensing when compared with ground survey methods is that it can be cost-effective over large areas and monitored efficiently by desktop analysis. In contrast, ground transects are field intensive and costly to survey over large areas. However, there are limitations with using remote sensors for ecological monitoring. Any mapping derived from remote sensing will only be as good as the input data—the scale and resolution of the mapped information is only as detailed as the input mapping used. The analysis and subsequent outputs are only relevant for the period of the data used and features can only be identified if active during the period of capture.

Although remote sensing can be used to map a wide area of land cover and vegetation from a surface perspective, it is important to validate the remote-sensing interpretations by ground truthing. This ensures that the realities on the ground match those observed from a digital sensor and that misrepresentation of surface features is minimised. In addition, field verification helps reduce the data and sampling requirements for monitoring processes and can provide useful information when preparing for broad coverage image analysis (e.g. identifying classes established by unsupervised classification, helping to select training sites for supervised classification). Field data also can be used to measure spectral profiles and collect physical properties of surface features that can be used to develop a spectral library, which will improve the quality of any subsequent image analysis because more information about the monitored phenomena will be known.

In practice, the area of ground-truthing sites is small compared with the remotely sensed area under image analysis. Also, the appropriate and representative sample for field validation would depend on a number of factors, including the variability in the landscape and the resolution of the remotely sensed image being analysed. Field verification becomes more important when large areas of imagery are analysed at medium spatial resolution (e.g. in the order of 100 m) because the ground covered by a pixel consists of several land cover types that fall within a single grid. Therefore, field data collected should be comparable with the spatial resolution of the image data (Baccini et al. 2007), to verify the contents of the pixel on the image.

6.2.1 Passive remote sensing: multispectral digital sensors

Successful use of remote-sensing data to monitor ecological change is based on a clear understanding of the relationship between ecological metrics and spectral signatures in surface images, the latter differing spatially and/or temporally from the surrounding areas. Ecological characteristics that are detectable by remote sensing can include changing vegetation community patterns, vegetation condition, boundary extent and erosion.

Image classification based on observing spectral and textural signatures from different vegetation communities can be used to indicate stability or changes in distribution of vegetation patterns. Temporal data, such as wet and dry imagery, can be used to establish natural variability in the landscape so that accelerated change caused by mining impacts—for example, subsidence induced by mining—can be identified.

Multispectral indices, such as the NDVI and normalised difference moisture index (NDMI), are techniques that can be used in image analysis. The NDVI indicates vegetation vigour, or greenness, on the assumption that high chlorophyll absorption in plants infers information on plant health, and the NDMI highlights water saturation or moisture content (Gandaseca et al. 2009; Segah et al. 2010).

The enhanced vegetation index (EVI) is a more recently developed remote-sensing technique that uses data gathered by a detection instrument of relatively finer resolution and relatively high frequency than is used by the NDVI. The EVI has several advantages over the NDVI. As suggested by Weier and Herring (2000), one advantage is the ability of the EVI to more accurately interpret remotely sensed data for vegetation coverage by incorporating corrections for atmospheric and soil influences. Also, the EVI incorporates information from the blue, red and near infrared (NIR) bands, whereas the NDVI includes only the red and NIR bands. Plant stress is indicated when plants have altered ratios of chlorophylls a and b, which differ in their reflectance spectra, especially in the blue band. Hence, the EVI is likely more sensitive than the NDVI due to changes in the plant pigments over a growing season.

The combination of these multispectral indices can be used, for example, to help differentiate bog from surrounding surface cover, such as sedge lands or grasslands, and identify any gain or loss in woody vegetation. This method is most effective if the phenology of the vegetation is documented and especially if changes in phenology due to presence or absence of water is known. As such, ideally an additional requirement would be supplementary detailed vegetation mapping and field verification to refine the image-processing outputs.

Monitoring vegetation condition within peat swamps is generally done by monitoring changes in vegetation distribution (Jenkins & Frazier 2010). Periodic mapping from field surveys in conjunction with high-resolution aerial photographs allows detailed changes of individual tree cover or groups of vegetation classes to be identified.

Field surveys (as described in Section 6.1.1.3) would be necessary to ground truth and verify the vegetation condition predicted in the image analysis.

Changes in bare ground, indicating peat cracking or collapse, or evidence of fire scars can also be delineated by visual interpretation of aerial or high-resolution satellite imagery. Mapping land cover using imagery to detect linear clusters of bare soil can be used to monitor erosion, but visual verification would be required to examine the proximity to a stream network to confirm gully erosion. However, since the geometry of gully erosion can be elongated and narrow, this method of mapping erosion is most effective where the extent of the impact is larger than the pixel size of the image and lends itself well to high-resolution aerial photography. The cost of high-resolution aerial photography is, however, high, particularly when repeat surveys over large areas are needed. More cost-effective satellite imagery—for example, 2 m multispectral imagery—offered by commercial vendors, such as GeoEye and WorldView-2, could deliver relatively similar information on land cover changes in vegetation distribution over time.

The current state of satellite remote sensing means that there is a trade-off between four factors: spatial detail, temporal detail, spectral detail (i.e. the number of different wavelength intervals in which measurements can be made) and cost. The current remote-sensing data sources used for ecological monitoring are the moderate-resolution instruments MODIS and Landsat TM, and high-resolution instruments such as GeoEye and WorldView-2.

The MODIS instrument provides moderate-resolution remote-sensing data at a spatial resolution of 500 m over Australia at a high-temporal resolution of eight-day intervals. CSIRO has a standardised approach to image pre-processing, including corrections for atmospheric effects for scenes over Australia (Paget & King 2008), and are made available for image analysis at no cost. However, MODIS may have limited application to monitoring temperate highland peat swamps because of its coarse resolution; many swamps in the Sydney Basin are smaller than 500 m wide.

Landsat 5 TM and Landsat 7 ETM+ have a resolution of 30 m, and data are captured over Australia bimonthly. Landsat TM instruments would be well suited for use in detecting seasonal patterns of environmental monitoring and for 1:25 000 mapping. Geoscience Australia holds the national long-term archive of Landsat data (1979–2013). Data from 1998 to 2012 has been processed and analysed to detect surface water, as part of the National Flood Risk Information Project.[4]The results are available online through the National Computational Infrastructure and the National Research Data Storage Infrastructure under the Creative Commons Attribution 3.0 Australia licence.[5]

Landsat has similar spectral information to the MODIS instrument, and has been used for a large range of vegetation applications using a variety of methods based on spectral and temporal signals. High-resolution data from both aerial photography and satellite platforms are an invaluable visual tool for field validation by ecologists. The multispectral imagery at 2 m resolution that is available for project-specific sites are useful in regional studies for detailed mapping of landscapes; however, these datasets become expensive when establishing baseline or monitoring studies.

High-resolution satellite imagery such as WorldView-2 or GeoEye is the best to use to monitor the peat swamps because of the small size of the swamps. Small changes within the peat swamps can be monitored. The high-resolution imagery would also allow vegetation communities to be monitored with higher detail compared with MODIS or Landsat. WorldView-2 and GeoEye also provide multispectral information, which facilitates the generation of environmental indices, such as the NDVI and the NDMI, to highlight vegetation vigour and moisture content, respectively. However, it can be difficult to capture high-resolution satellite imagery data within an ideal monitoring timeframe because of commercial priorities around new data acquisitions. For example, despite historical data being available since the early 2000s (e.g. Ikonos at 4 m resolution and QuickBird at 2.4 m), archive availability of multispectral data is relatively low. This is because commercial sensors predominantly capturing data based on commercial orders and in high-interest areas; this

[4] www.ga.gov.au/hazards/flood/floods.html
[5] http://creativecommons.org/licenses/by/3.0/au/deed.en

contrasts with Earth observation programmes, such as Landsat and MODIS, which continuously capture global data.

Data captured by commercial sensors will require pre-processing for radiometry, such as atmospheric corrections, before image analysis for land surface biophysical variables (e.g. vegetation condition) or geophysical variables (e.g. vegetation extent) can start (Lillesand et al. 2008). This is especially important when the area of interest is to be monitored over time or over a large extent.

6.2.2 Active remote sensing: lidar and radar

Lidar is an active-sensing system rather than a passive sensing system, because a laser is emitted; it is therefore less affected by weather than optical remote sensing. Multiple ground reflections can occur from a single laser pulse so that modelling can be based on the assumption that the first reflection represents a partially transparent surface (generally the canopy) and the last represents the lowest hard surface, which could be the ground or a dense canopy. There is typically one point of measurement per square metre with a 0.15–0.3 m vertical accuracy. A complete description of lidar is provided in Wehr and Lohr (1999).

As lidar can differentiate between bare earth and ‘non-ground’ points such as vegetation, buildings, bridges, water (ICSM 2010), potential erosion can be identified based on the classification of bare earth lidar pulse returns. For a more detailed delineation of gully erosion, the lidar dataset can be used to create a dense elevation model from which a stream network can be used to locate bare ground areas on gradient slopes. This would provide a comparatively more detailed delineation of gully erosion than land cover mapping from imagery because a stream network is a more reliable input. The spatial coverage of lidar can be up to four or five elevation points per metre, producing a dense dataset. In addition, lidar considers the topographic component of the environment by using gradient information to better inform feature identification. However, this method would be more costly in data acquisition for temporal analysis unless the data were available for multiple uses (such as canopy height modelling or investigation into the vertical distribution of foliage).

Synthetic aperture radar (SAR) is an active sensor system emitting radar signals. Unlike lidar, which is subject to visibility requirements, SAR can operate in any weather conditions, day or night. Depending upon the particular system design, SAR emits a variety of wavelengths, some of which can penetrate vegetation (Bamler & Hartl 1998). Radar paths are oblique to the vertical, to the extent that reflected rays are scattered off calm water and thus there is no return signal. Points are measured at intervals of 2–10 m with a vertical accuracy 0.5–5 m.

Vegetation structure, canopy density and understorey condition can be modelled using either lidar or radar operating in X- and P-band frequency modes. Both technologies allow the development of both a digital elevation model of the terrain and a surface model at the canopy level. The difference between the two layers generates a canopy height model that can be used to separate swampland from fringing vegetation and delineate boundary extent (Jenkins & Frazier 2010). This can give an indication of changing swamp extent in response to fluctuations in inundation. A comparison of elevation models generated using SAR data is given in Hoja and d’Angelo (2009).

Although land cover mapping and changes in community pattern from multispectral imagery can be used to monitor swamp extent, there can be cases where overhead obstructions, such as dense canopy cover, prevent accurate depiction of boundaries from an aerial view. In these cases, SAR can be used to detect changes in extent. Areas exemplifying high soil moisture can be a measure to indicate an active green swamp and can be observed by an L-band radar system (Hoekman & Vissers 2007).Satellite radar is often used to monitor peat swamp forests in tropical environments, because the technology is well suited to environments obstructed by cloud cover. However, when applied to monitoring peat swamps over the Sydney Basin where atmospherics are not a persistent issue, satellite radar may be limited by the resolution offered by current satellite systems, such as ALOS PALSAR and RADARSAT (at around 10–100 m resolution, depending on imaging mode). Higher resolutions are becoming increasingly available from recently launched constellations such as COSMO-SkyMed; however, its application for environmental use has been limited.

Vegetation biomass can be monitored using C- and L-band SAR based on a combination of polarimetric and interferometric analysis (Pol-InSAR) to map parameters such as canopy cover and soil moisture. However, using Pol-InSAR to monitor vegetation biomass is in its infancy, and more investigation is required on the interactions between the radar sensor and natural surfaces (Lu et al. 2007). DInSAR (differential interferometry) is usually used to monitor changes in vertical movement and is therefore more applicable to monitoring changes in topography caused by subsidence, rather than ecological responses, as discussed in Section 6.3.2.

6.3 Monitoring cracking and tilting of the sandstone surface

The geotechnical monitoring methods outlined in this section assume that the mining-induced subsidence wave, advancing beneath a peat swamp at the rate of up to 10 m/day, causes the sandstone substrate to stretch; then tilt, subside by 1–2 m and become level again; and finally to recompress. This cycle of subsidence forces joints in the sandstone to open, then close—but the final fit is not as close as it was originally. In addition, some new fractures are created and some sliding occurs along bedding planes. These rock mass disturbances are presumed to extend to depths of 10–15 m below the surface.

The objective of geotechnical monitoring is to identify cracking of the sandstone beneath, or in the vicinity of, the swamp. Any indication of cracking suggests that the swamp is at risk of impacts (either through changes to the underlying sandstone itself or from associated changes to the swamp hydrology) and that adaptive management measures should be triggered.

A geotechnical monitoring programme in this environment should:

- map the extent, orientation, aperture and distribution of the subsidence-induced fractures in the near-surface sandstone
- measure, where possible, dilation of joints and bedding planes to a depth of 10 to 15 m and the resulting changes in permeability of the rock mass
- document any other changes to the rock mass observed during drilling and borehole testing.

The most severe subsidence impacts can be expected close to longwall panel edges, close to cliff lines, and on the floor of steep-sided gorges. Of the three conceptual models, hanging swamps and valley infill swamps are therefore most vulnerable to subsidence impacts. Monitoring should identify swamps at highest risk of impact and design a monitoring programme that focuses on these swamps, while recognising the need for control sites (i.e. the before–after control–impact approach). Most of the subsidence damage to the rock mass occurs relatively quickly after the longwall face passes beneath, in about one week. After a few weeks, subsidence impacts stop until the adjacent longwall panel is mined, usually about one year later. Monitoring of rock deformation should therefore continue for 18 months following panel extraction.

The capabilities and shortcomings of various geotechnical monitoring methods are summarised below. Some are in routine use, some have promise for the future, and some may be useful at one site and not another. All of the methods discussed require site access, which can be difficult in the highly vegetated areas where the peat swamps occur since vehicle access for large equipment is required. Because methods are site specific, a mining lease that contains many swamps will require a large volume of geotechnical work to characterise the swamps and to monitor for impacts.

A subsidence monitoring programme for a single swamp should include three-dimensional (3D) surveys and a geotechnical drilling programme that includes borehole coring, logging and permeability testing. Further information on monitoring of subsidence effects is available in SKM (2013) and CoA (2014a).

6.3.1 3D survey methods

Conventional 3D survey monitoring of pegged lines provides the essential background for all subsidence investigations, and is now a requirement over all longwall mining panels in New South Wales. Survey lines are laid out along the panel centreline and along one or several lines perpendicular and oblique to this. One or more of these lines can be routed through a swamp. The peg spacing is typically 10 to 20 m, but can be reduced to 5 m for more detail. Horizontal movement of the pegs is measured using a laser theodolite or a GPS. Coordinate (X, Y, Z) accuracy is within a few millimetres, subject to line-of-sight constraints. This method is also useful for detecting far-field ground surface movements 1 to 2 km from the swamp site.

In peat swamps, 3D surveys can provide the overall subsidence profile intermittently (say one survey round each 4 to 6 weeks during panel extraction). The most relevant information for identifying potential impacts on peat swamps is measurement (in mm/m) of lateral movement, or ground strains (average stretching and compression between pegs). Transient tilts can likewise be derived from the survey data. Some difficulty with lines of sight and peg stability can be expected within swamps.

6.3.2 Remote sensing

Airborne and satellite height measurements (lidar and SAR) offer future promise for monitoring subsidence effects such as cracking and tilting of the sandstone or swamp surface. Traditionally, they have been too expensive and largely inaccessible to the Australian market, but at present are becoming more accessible and readily available. The advantage of monitoring via remote sensing is the economies of scale, with it being more economical to use remote sensing over large areas than traditional field surveys.

Small deformations such as cracking or tilting of a sandstone surface are likely to be more difficult to detect by remote-sensing technology. Research into InSAR methods states that millimetre precision is detectable as a relative change (Chang et al. 2009). The shorter the wavelength of the radar, the more sensitive to vegetation cover terrain measurement will be (e.g. X band at 2.5 to 4 cm, C band at 4 to 8 cm) and the longer the wavelength, the more likely the radar is to penetrate vegetation and observe ground measurements (e.g. L band 15 to 30 cm, P band 74 to 100 cm). When using SAR (either X, C, L or P bands) for DInSAR analysis to monitor changes in terrain, the accuracy of the analysis is largely dependent on the quality of the input data. Image geometry and temporal conditions play a large role in the quality of the output results, as discussed in Chang et al. (2004).

6.3.3 Geotechnical methods

6.3.3.1 Borehole testing and monitoring

A standard truck-mounted drilling rig should be able to set up and drill a 15 m borehole in a day. Other drill mountings on tracked all-terrain vehicles could match this, but at higher cost. Small self-propelled and even man-portable drilling rigs are much slower and more expensive per metre drilled. The last resort, for very high-priority boreholes, would be helicopter transfer of the rig, in several lifts.

The following monitoring techniques are suitable for use in 75 to 100 mm diameter cored boreholes:

- Core logging. Careful logging and photographing of a 50 to 60 mm diameter sandstone core recovered from the borehole will yield details of bedding planes and rock fabric variations, but not of most joints because these are usually subparallel to vertical holes. Fracture spacing measurements on drill cores usually overstate the degree of fracturing, since many of these breaks are drilling induced.
- Borehole camera. Borehole cameras, such as the RAAX system, give a better picture of rock mass fracturing (including orientation, precise depth and aperture) than core logging. However, the two methods are better thought of as being complementary rather than alternatives. In addition, the borehole camera requires a hole filled with clean water.
- Water injection (Lugeon) permeability tests. These tests measure the degree of fracturing over a 2 to 3 m interval in terms of water loss under standard conditions, preferably before and after subsidence. For best results, a second borehole should be drilled and tested after the subsidence wave has passed, since pre-existing fractures may have become clogged. The packer assembly is about 4 m long and has a headroom problem—it cannot get results at depths less than about 3 m.
- Downhole geophysical logging. Described in Section 6.3.4, these are to a certain extent complementary to the borehole camera and the Lugeon testing. The preferred methods for sandstone beneath peat swamp sites are sonic and neutron.
- Extensometer. These come in several forms (wire, rod, tube-mounted magnets), but all measure vertical movements in a borehole between fixed points (anchors) to an accuracy of about 1 mm. They are therefore very useful for detecting delamination (cracking) along bedding planes. An extensometer in a 15 m deep borehole might have 5 to 6 anchors at intervals of 2 to 3 m. The aperture (opening and closing) of fractures within these intervals is the measurable parameter, not their precise depth. An extensometer can be thought of as being complementary to Lugeon water pressure tests—the former records crack opening, the latter measures water losses into these cracks.
- Inclinometers, stressmeters and tiltmeters. These are devices installed in boreholes to record ground movements and stress build-up during the passage of a subsidence wave. However, they yield no specific information on ground cracking and so are not recommended.

6.3.3.2 Joint monitoring

Joint mapping can record the opening and closing of joints and the creation of fresh fractures within the sandstone rock mass. If joints open up beneath the swamp, moisture may drain through its base into the underlying sandstone. Identification of joints in the swamp area (as monitoring cannot occur beneath the swamp) can indicate heightened risk of impacts to the swamp. To successfully identify joints, several things are required:

- access to the swamp site during the critical few days of extension, tilting and ground lowering when the longwall face passes beneath at the rate of perhaps 5 to 10 m per day. It is desirable that a return visit be made after the compression phase has largely been completed, several weeks later
- a good base map at about 1:100 and possibly an enlarged aerial photo with some surveyed control points on it. Good rock exposure is desirable, but even under the most favourable conditions this may amount to only 10 per cent of the study area
- a network of monitoring points (roofing nails in the rock) and tell-tales (glass slides glued across natural cracks) over which stretching and compression can be measured to an accuracy of about 1 to 2 mm. Home-made feeler gauges and calipers for measuring joint opening and closing may also be useful.

A range of more precise aperture measurement devices (crackmeters, reading to less than 1 mm) is available, but these are more costly. Joint mapping techniques for geotechnical purposes are dealt with in some detail by Palmstrom (2001).

6.3.4 Geophysical methods

A number of geophysical methods have been suggested for monitoring the effects of subsidence near swamps. The main shortcoming with these is that they do not directly measure any properties that indicate impacts—such as rock mass permeability or crack development. By far the most useful are the downhole logging tools, though the 3D televiewer could be applicable. The methods are as follows:

- Downhole sonic logging. This measures changes in ultrasonic wave velocity in borehole walls, both in detail (short-spaced array) and in bulk (long-spaced array). Low velocities are correlated with porous or fractured rock. At its best, the sonic tool can pick up individual bedding plane fractures and resolve to about 50 mm in depth. However, it must be run in a water-filled borehole and requires a head room of 2 to 3 m to operate (McNally 1987).
- Downhole neutron logging. The neutron log responds to hydrogen ion content, and hence to water content and porosity in most rocks (such as sandstone). Once again a water-filled borehole is required and no readings are possible for the top 2 to 3 m. Resolution is not quite as good as the sonic logging tool, but both can be run at the same time for more or less the same price (McNally 1987).
- Borehole 3D televiewer. This is a more sophisticated tool than the sonic logger, combining features of a borehole camera with full waveform recording to generate a synthetic picture of wall rock conditions (Paillet 1985). Though it has much to offer in assessing fractured rock masses, there appears to be no published information on its use in shallow boreholes in Australia. However, it can operate in a muddy borehole that would defeat a borehole camera. The televiewer pulses ultrasonic energy into the borehole wall, with the degree of wave attenuation recorded being proportional to the degree of fracturing encountered. The results are presented in a 360° image of the borehole wall, with crack locations (along with their orientation and aperture) interpreted from the variations in wave amplitude.

- Electromagnetic conductivity (EM). EM measures changes in, among other things, soil moisture content to a depth of several metres, but without sufficient precision to detect individual fractures in a sandstone rock mass (Milsom 1989; McNeill 1997). Its main selling point in swamp investigations is that it can be operated on foot by two people or, if access permits, by one person on a quad bike. It is therefore less invasive than borehole methods and offers a continuous profiling capability. Because the equipment is light it allows easier access to the swamps. However, results are less reliable than the downhole methods.
- Ground probing radar (GPR). GPR also measures changes in ground conductivity, and hence moisture content to a few metres depth, but with more precision and at greater cost than EM. The depth imaged varies with the pulse frequency: high frequencies (short wavelengths) give improved resolution, but reduced penetration. Although GPR gives no specific information on fracture locations, it might be useful for assessing changes in peat moisture content. Two shortcomings may limit its application in peat swamps: it requires good contact with the ground being probed, which should be firm and even, and it performs best in dry ground. The equipment is typically operated on a sled, a motorised wheelbarrow or similar device (Davis & Annan 1989; Van Overmeeren 1997).

6.4 Monitoring changes to swamp hydrology

Cracking and tilting of the sandstone beneath peat swamps can result in altered groundwater flow regimes near the swamp, which may cause the regional or perched aquifer in the sandstone to become disconnected from the swamp, ultimately resulting in reduced groundwater discharge to the swamp and lower or less frequent levels of inundation. The fractures in the sandstone at the base of the swamp may also cause water to drain out of the peat layers, which also leads to reduced inundation. These changes in groundwater flow regimes can be identified through monitoring the shallow groundwater, including water levels within the sandstone aquifer and within the peat itself. Bayesian belief network modelling supported the understanding that swamp ecology was highly sensitive to changes in inundation within the peat. Methods for monitoring shallow groundwater impacts are discussed in Section 6.4.1. Deep groundwater monitoring (discussed briefly in Section 6.4.2) is less relevant, as near-surface impacts present the highest risk for peat swamps.

Other hydrological impacts associated with longwall mining include an altered flow regime for surface water flowing through the swamp, and water quality impacts associated with waste water discharge uphill from the swamps. An altered surface water flow regime is a direct result of subsidence and may cause an increased velocity of flow, which heightens the risk of peat erosion. These potential impacts can be monitored through surface water monitoring methods, discussed in Section 6.4.4. The water quality monitoring appropriate for monitoring mine waste water discharge quality and potential impacts on the peat swamps is discussed in Section 6.4.5.

A hydrological monitoring programme primarily aims to identify changes in local groundwater levels caused by the flow of groundwater into subsidence-induced fractures, changes in the level of inundation within the peat and change in surface inflow quantity and quality to the swamps. Monitoring before, during and after mining is important to understand the natural hydrological variability and to be able to identify changes caused by longwall mining impacts.

6.4.1 Shallow groundwater monitoring

Monitoring to identify potential impacts on swamps should focus on shallow groundwater impacts because these provide the most direct indication of follow-on impacts to swamp ecology. Fracturing and cracking of the sandstone as a result of subsidence can impact swamp inundation in two ways:

- by increasing the permeability of the underlying sandstone aquifer and causing the connected aquifer to become less connected (or disconnected) from the base of the peat swamps. This is more likely to occur in valley infill and hanging swamps since they are most frequently connected to perched or regional aquifers within the sandstone
- by increasing the permeability of the unsaturated sandstone and causing moisture held within the peat sediments to drain through the base of the swamp. This could occur in all types of peat swamps.

Monitoring water levels within the peat sediments should occur at all swamps. To directly monitor changes in water levels within the peat swamp sediments, shallow piezometers.(a water bore used specifically to measure groundwater head and/or quality) should be installed into the peat and monitored for at least two years before mining. These bores may be able to be installed by a hand auger, thereby removing the need for large and invasive equipment on the surface of the swamps.

Piezometers should also be installed into the sandstone around the swamps to determine whether the swamp relies on a connected groundwater resource to maintain inundation. With a focus on valley infill swamps (as the most at risk from subsidence impacts due to their position in steeper topography), piezometers should then be drilled into the sandstone uphill and downhill from each swamp and monitored for at least two years before mining. This gives baseline information that indicates whether groundwater is likely to be connected to the swamps and data on the temporal variations in connection. Permeability testing using pumping tests or slug tests should also occur on these boreholes to give information on the natural sandstone permeability before mining. Permeability testing should be repeated at intervals during and after mining to identify any changes.

Two years is considered the minimum period necessary for baseline monitoring, as recommended by the Southern Coalfield Inquiry (NSW DP 2008), Metropolitan Inquiry (NSW PAC 2009) and NSW OEH (2012b). Ideally the length of baseline monitoring would extend for as long as needed to establish the range of natural variability in the swamp systems.

If groundwater in the surrounding sandstone is shallow (say 10 m below groundwater level) at either of the bores, it should be assumed that interaction between the swamp and the aquifer exists, and any cracking or fracturing at the surface of the sandstone therefore has the potential to change swamp inundation. If the groundwater is deep, long-term swamp monitoring should focus on changes in water levels within the peat.

At least a week after installation, groundwater quality from piezometers installed in both the sandstone and the peat should be tested. Parameters for analysis should include electrical conductivity, pH, acidity, alkalinity, nutrients, dissolved oxygen, major cations and anions, microorganisms, turbidity, methane and metals (including iron and manganese).

Groundwater and geotechnical baseline monitoring programmes should be integrated because most geotechnical methods can be done in drilled holes before the piezometer is installed. The specific methods mentioned in this section for monitoring shallow groundwater are discussed in more detail in Section 6.4.3.

6.4.2 Deep groundwater level or pressure monitoring

Deep boreholes that monitor groundwater in the regional aquifer are less likely to provide useful information since most peat swamps are not connected to the regional aquifer—although valley infill swamps may be connected to regional aquifers (see the conceptual models in Section 3.1). The other problem with installing and monitoring deeper piezometers is their susceptibility to damage caused by rock deformation at greater depths. Monitoring changes to near-surface groundwater flow provides a more direct and reliable indication of changes in groundwater levels that may ultimately impact the peat swamp ecology.

However, monitoring of groundwater inflows to longwall panels can be used to provide an early indication of potential groundwater impacts for swamps that interact with the regional aquifer. If more groundwater is flowing into longwall panels than was predicted by modelling, it can be assumed that the risk of groundwater impacts (such as change to flow paths, lower watertables in the regional aquifer, decreased connection with perched aquifers or surface features) is higher.

6.4.3 Groundwater monitoring methods

The methods for monitoring groundwater impacts on swamps are largely limited to direct measurement of water levels through piezometer monitoring. As with the geotechnical monitoring methods described in Section 6.3, the main obstacle to installing a comprehensive groundwater monitoring programme is site access, since road access is generally required to install drilling equipment. Installation of piezometers in the swamp sediments is easier than into the underlying or adjacent sandstone, because the boreholes could be relatively shallow and could be installed using a hand auger.

6.4.3.1 Piezometer installation

Monitoring groundwater involves drilling and installing screened piezometers in aquifers (see example shown in Figure 6.1). The water levels measured in piezometers represent the average head at the screen of the piezometer.

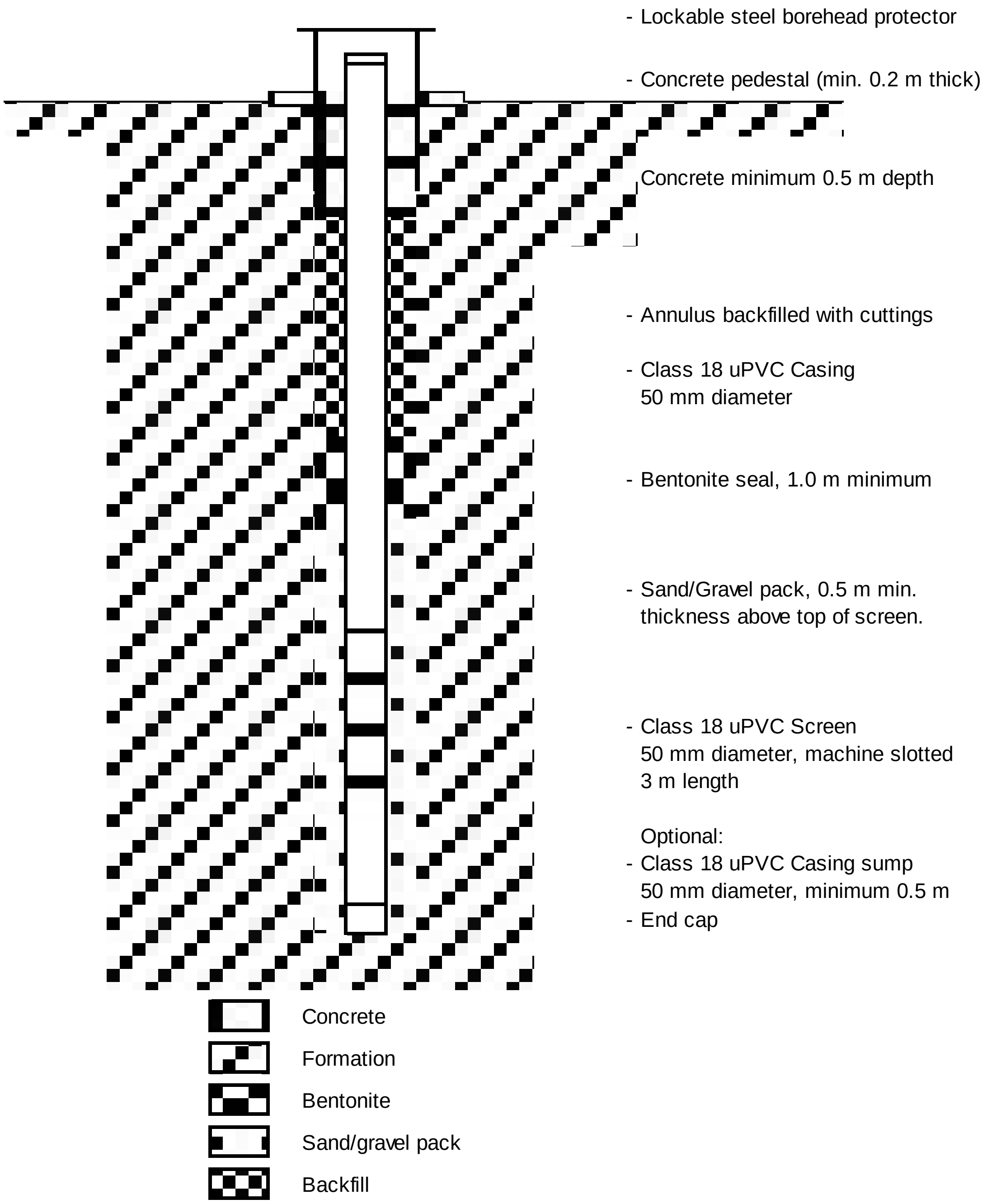

Figure 6.1 Example of a piezometer used to monitor groundwater levels and water quality.

To ensure that deep piezometers are monitoring the correct aquifer, they need to be hydraulically isolated from overlying or underlying formations, which can be achieved through appropriate bore construction. All piezometers and bores should be constructed according to *Minimum construction requirements for water bores in Australia* (NUDLC 2012). If a large diameter hole is drilled, multiple piezometers can be installed at different depths to allow water levels to be monitored in different aquifers. Multiple piezometers at different depths at the same site are often used to determine differences in groundwater pressures between overlying aquifers, to identify vertical hydraulic gradients and potential for vertical groundwater movement. Subsidence usually results in varying piezometric responses because the mining face approaches the monitoring piezometer (Hua et al. 2012).

The use of piezometers to monitor potential impacts to swamps is the preferred method because it allows physical measurement of shallow water levels and the collection of groundwater samples for water quality analysis. Shallow groundwater monitoring provides the most reliable indication of potential impacts to peat swamps because most peat swamps and perched aquifers are not connected to the deeper, regional aquifer and therefore will not be impacted by changes in groundwater pressure or flow regime of the regional aquifer. Therefore, some of the problems associated with installation of deeper piezometers can be largely avoided—for example, inter-aquifer leakage through the borehole and the need for very large equipment. Only where connection exists between the peat swamp and the regional aquifer (e.g. in some valley infill swamps) should deep groundwater bores be installed to monitor the regional aquifer.

6.4.3.2 Piezometer water level/pressure monitoring

Pressure transducers connected to data loggers are often used in piezometers to provide an almost continuous record of groundwater pressure data. The transducers are installed below the water level in each piezometer. Ideally, a barometric pressure transducer would also be installed above the water level in one of the swamps in the monitoring area. The data loggers connected to the transducers can be set to record at regular intervals or at specified groundwater pressure changes. Transducers are the preferred method for monitoring groundwater levels since they allow frequent recording of groundwater levels and require relatively infrequent site visits, which reduces the costs of data collection.

Alternatively, groundwater pressures can be monitored manually using fox whistles (devices designed to locate the water level in a bore through an audible response), electric tapes (devices designed to locate the water level in a bore through the completion of an electric circuit as electrodes become submerged) or pressure gauges connected to the wellhead, if the aquifer exhibits artesian characteristics (groundwater heads that are above the ground surface). These methods are manual and therefore require staff to physically measure the water levels. This generally results in less frequent measurements being taken and higher monitoring costs in the long term.

6.4.3.3 Vibrating wire piezometers

An alternative to a conventional screened piezometer is a vibrating wire piezometer that converts a water pressure to a frequency signal via a diaphragm, a tensioned steel wire and an electromagnetic coil. The piezometer is designed so that a change in pressure on the diaphragm causes a change in tension of the wire. When excited by the electromagnetic coil, the wire vibrates at its natural frequency. The vibration of the wire in the proximity of the coil generates a frequency signal that is transmitted to a readout device at the surface. Vibrating wire piezometers can be directly grouted in using a bentonite–cement grout allowing multiple piezometers to be installed at different levels in the same borehole.

Vibrating wire piezometers are widely used in the mining industry and are best suited to situations where groundwater pressure information is required from multiple depths since the same hole can be used for monitoring pressure at varying depths. They are not appropriate if information on groundwater quality is required, as they are grouted up to the surface and therefore groundwater cannot be sampled.

Vibrating wire piezometer strings grouted into the borehole can sometimes stop operating because they can snap off sequentially behind the face. However, the cessation of operation is sometimes used to track the rising height of caving and rock mass disturbance as mining proceeds.

6.4.3.4 Permeability testing

The key aquifer hydraulic characteristics that characterise a geological unit's ability to store and transmit groundwater are:

- hydraulic conductivity—describes the flow velocity of water moving through a porous medium under a unit gradient of hydraulic head (Heath 1983), expressed as distance over time (e.g. m/day). Horizontal hydraulic conductivity (Kh) refers to the flow in the horizontal direction whereas vertical hydraulic conductivity (Kv) refers to flow in the vertical direction
- storage coefficient—describes the amount of water that a unit volume of saturated permeable rock would yield if drained by gravity (Richardson 2011).

Longwall coalmining can cause cracking and fracturing near the surface of the sandstone, which impact on the hydraulic conductivity and storage coefficient, and therefore on the capacity of the aquifer to store, transmit and yield water. Monitoring changes in aquifer hydraulic parameters is important to test the assumptions used in groundwater modelling predictions conducted pre-mining and as inputs to post-mining groundwater modelling.

In-hole rock mass permeability (Lugeon) testing, before and after mining, has been used as a measure of mining-induced fracturing (hard rock aquifers only) and changes to aquifer permeability since the mid-1970s. The method involves injecting water into a section of the borehole sealed off above and below by inflatable packers. Usually the test section is 3 or 6 m long. Large water losses indicate the presence of one or more open fractures within the test interval. One drawback of the test is that, usually, the equipment cannot maintain sufficient flow when the fractures are more than a few millimetres wide, whereas some subsidence delaminations can be more than 200 mm wide. Nevertheless, a water loss beyond the capacity of the pump is, in itself, an important finding.

Monitored changes in waterlevel over time and the rate of water injection can be used in various pumping test solutions (e.g. Theis 1940) to calculate the aquifer permeability. The resultant permeability is expressed as a hydraulic conductivity or transmissivity (hydraulic conductivity multiplied by the aquifer thickness). An estimate of the aquifer storage coefficient can only be made if water level information is monitored in non-pumped observation bores during the test. In most situations, Lugeon tests do not include observations in monitoring boreholes, so the aquifer storage coefficient cannot be calculated.

Lugeon testing for testing fracturing in hard rock aquifers may be supplemented by borehole camera surveys. Under ideal conditions, with the borehole filled with clear water, the location, attitude and aperture of each fracture—whether a natural joint or a subsidence crack—can be recorded. Unfortunately, this technique is more expensive than water injection and the device cannot be used in a dry hole or where there is any risk of it jamming in the bore during deployment.

In addition to Lugeon testing, other pumping test techniques are available to determine aquifer hydraulic properties although they are not commonly used in longwall mining. For example, temporary pumps can be installed in bores and used to pump water out at a constant rate while monitoring the changes in water level in the pumped bore. As with Lugeon testing, various pumping test solutions are available to calculate the hydraulic conductivity of the aquifer (see CSIRO & SKM 2012 for more details).

6.4.3.5 Groundwater quality testing

No available literature suggests that subsidence has caused impacts to water quality which has then impacted swamp ecology. However, subsidence-related impacts to streams in the

Sydney Basin have been widely documented. If these impacts occurred high in the catchment, it is likely that downstream swamps would also be impacted. Potential impacts to stream water quality include increased metal content (iron and manganese), formation of algal mats, and methane bubbles in standing water. These changes in water quality could affect both surface water flowing into the swamps and shallow groundwater in the vicinity of the swamps.

Water quality impacts resulting from mine waste water discharges may include increased salinity, with a higher content of sodium, potassium, calcium, manganese, chloride, sulfate and bicarbonate (ACARP 2002), and iron and zinc (ACARP 2000). Mine waste water releases also have the potential to contaminate shallow groundwater.

When sampling groundwater piezometers, it is vital to ensure a representative sample of the aquifer is obtained. The two accepted methods for sampling groundwater in Australia are (Sundaram et al. 2009):

- purging—the stagnant or standing water resident in the borehole is removed from the bore, drawing water through the screens from the aquifer. The generally accepted standard is for three casing volumes to be expelled before a representative sample is taken.
- low flow—formation water is extracted through the bore screen (or screened interval) at a low rate, without disturbing the stagnant water column above. This is achieved by pumping at a rate that results in minimal drawdown of the water level within the bore. The sampling time is much less than the time required for the purging method. Typical flow rates for low flow sampling are about 1 to 2 L/min.

Samples should be transported appropriately (usually in chilled containers) to a laboratory accredited for the analysis. Suggested testing for shallow groundwater includes electrical conductivity, pH, acidity, alkalinity, nutrients, dissolved oxygen, major cations and anions, microorganisms, turbidity, methane and metals (including iron and manganese).

Changes in the chemical signature of groundwater samples can be used to determine whether mixing of waters from discrete aquifers has occurred. The ratios of various cations and anions in any one aquifer are often reasonably constant. Any changes in these ratios can signal mixing with waters originating from previously disconnected aquifers.

One method to highlight whether mixing between aquifers has occurred is to use a Piper diagram (Figure 6.2) to plot the ratios of various cations and anions (calcium, magnesium, sodium, potassium, sulfate, chloride, carbonate and hydrogen carbonate). Piper diagrams allow large numbers of analyses to be represented on a single graph so that trends or groupings in the data can be observed (Freeze & Cherry 1979). The cation/anion concentrations are plotted as percentages and, because of this, groundwater with very different total concentrations can plot in the same location on the diagram (Freeze & Cherry 1979). If leakage between two aquifers of different concentrations occurs (e.g. the peat sediments and the sandstone aquifer), in the Piper diagram, the cation/anion concentration in the resulting sample will plot on a straight line between the original two samples. Such an analysis requires samples to be collected and analysed pre-mining to provide the baseline chemistry for comparison with post-mining results. Ideally, a number of samples collected from piezometers installed into the connected sandstone aquifer (if it exists) and the peat sediments are required to develop meaningful results.

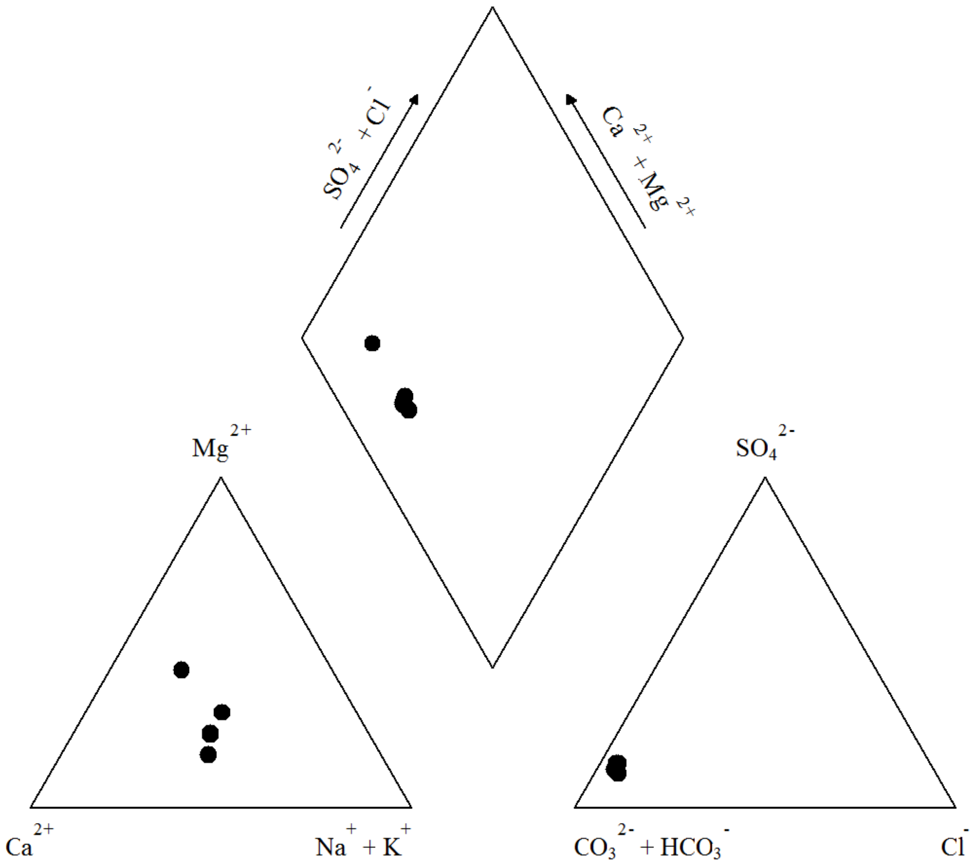

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Figure 6.2 Example of water chemistry results plotted on a Piper diagram; samples taken from catchments of the Mzingwane and Thuli rivers, Zimbabwe.

6.4.4 Surface water monitoring

Longwall mining can affect surface water flow and quality into the peat swamps by:

- causing cracking of the sandstone that intercepts streams above the swamps and reduces flow from headwater streams into the swamps
- causing cracking of the sandstone surrounding the swamps, which intercepts surface run-off after rain.

The latter of these mechanisms is unlikely, since cracking in the surrounding sandstone would have to be very extensive to significantly reduce run-off. Interception of streamflow above the swamps is somewhat more likely, particularly for valley infill swamps, although most streams that feed into swamps are low flowing and seasonal. Any permanent headwater streams will be groundwater fed and hence impacts would be more efficiently detected by shallow groundwater monitoring up-gradient of the swamps, as described in Section 6.4.1.

Monitoring of water quantity should include information such as:

- stream flow gauging
- climate data, rainfall, evaporation, temperature and evapotranspiration (for ungauged catchments and yield comparisons)

- stream substrate examination (for cracking or clogging).

Climate data is collected at weather stations located in the region of the mine site within the area of impact. Climate and water information are also available from the Bureau of Meteorology,[6] including river flow and flooding records and warnings, although there are few gauges at the high elevations of the peat swamps.

Many methods can be used for stream gauging (dilution, pressure transducers, Acoustic Doppler probes, flow meters, sphere release, weirs) and they can be highly variable in accuracy (Soupir et al. 2009). Stream gauging should be completed more frequently if the substrate is mobile. Sand substrates and streams with medium to high bed-load transport levels should be check gauged at least annually. Selecting natural features such as hard rock bars for measurement of flow is a useful approach to reducing the need for frequent field gauging.

6.4.5 Monitoring waste water discharge upstream of the swamps

It is important to monitor the water quality of streams near mining operations. Although recommendations on the nature of water quality monitoring vary, it is a priority to monitor the water quality of streams specific to a mining area and develop site-specific trigger values for key water quality parameters. Quality should be monitored:

- of water in headwater streams above wastewater discharge points
- of wastewater discharged from the mines
- of water downstream of discharges before it enters the swamps.

As headwater streams provide good-quality water for irrigation and domestic use, habitat for fish and other aquatic biota, and support other values such as recreation and aesthetics, it is important that the streams are monitored so that these values can be protected. Default trigger values for a range of water quality parameters to ensure protection of waterways are provided in *Australian and New Zealand guidelines for fresh and marine water quality* (ANZECC & ARMCANZ 2000a). The key indicators and their trigger values are provided in Table 6.3.

Table 6.3 Default trigger values for protection of aquatic ecosystems for slightly disturbed upland rivers.

Total phosphorus	Filterable reactive phosphorus	Total nitrogen	Oxidised nitrogen	Ammonium	Dissolved oxygen (% saturation)	pH	Turbidity (NTU)	Electrical conductivity (µS/cm)
20 µg/L	15 µg/L	250 µg/L	15 µg/L	13 µg/L	90–110	6.5–8	2–25	30–350

NTU = nephelometric turbidity unit

The guidelines also provide a series of trigger values for toxicants in fresh water at alternative levels of protection.

Where headwater streams are also used for domestic use, values in *Australian drinking water guidelines* (NHMRC & NRMMC 2011) should be applied.

[6] www.bom.gov.au

Because the key water quality issues associated with mine water discharge relate to elevated turbidity, chemical pollution, elevated metal concentrations and acidic conditions, the minimum water quality parameters that should be measured are:

- turbidity or total suspended solids
- trace elements (e.g. aluminium, antimony, arsenic, barium, beryllium, boron, cadmium, calcium, chromium, cobalt, copper, iron, lithium, magnesium, manganese, molybdenum, nickel, phosphorus, potassium, selenium, silicon, silver, sodium, strontium, tin, titanium, vanadium, zinc)
- oil
- pH and alkalinity
- electrical conductivity, salinity or bicarbonate levels
- major ions
- nutrients (total nitrogen, nitric oxide, nitrogen dioxide, ammonia, total phosphorus, filterable reactive phosphorus).

Monitoring should be done at various spatial and temporal scales. Spatially, sites should be located in headwater streams upstream of any discharge points, to provide a reference condition for downstream water quality. Sites should also be located immediately downstream of the discharge point, at appropriate points before the discharge waters enter the swamps and within the swamps themselves. Monitoring should occur at regular intervals and under various conditions—for example, following wet weather or when mine water is being discharged. Any seasonality associated with peat swamp plant community growth and recruitment should also be recorded and considered when analysing the data for significance. When a comprehensive dataset is obtained, this can be used to derive site-specific trigger values. More information on this can be found in the ANZECC and ARMCANZ (2000a) guidelines. These site-specific trigger values should be based on the environmental values of the stream and swamp, and may also be based on water quality measured at the upstream reference locations.

Macroinvertebrate surveys can also be used to measure the ecological response to changes in water quality and habitat. The metrics used to indicate macroinvertebrate health (as a response to changes in habitat and water quality) include AusRivAS (Turak & Waddell 2002), SIGNAL (Chessman 2003) and POET. Any changes in response from the baseline to the ongoing impact monitoring phases could be used to suggest an ecological response.

7 Ecological monitoring and reporting approach

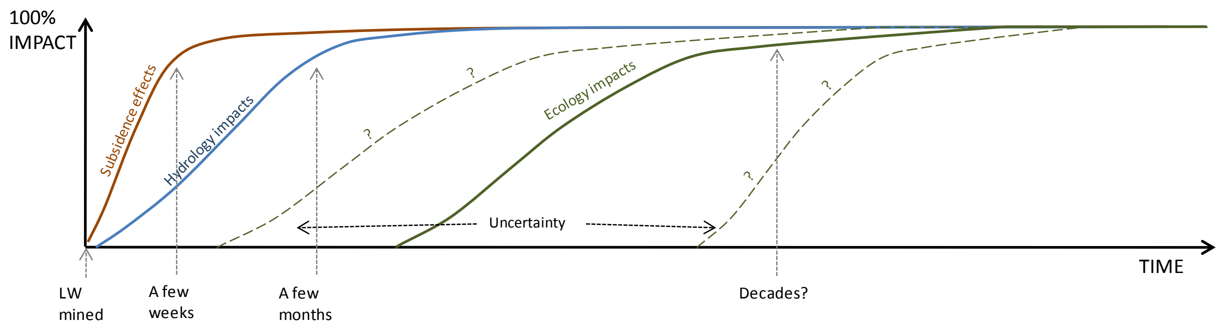
7.1 Scope of monitoring programme approach

To design an appropriate monitoring programme for detecting the impacts of longwall mining on peat swamps, the risks (potential impact pathways) must be clearly identified. This requires an understanding of the mechanisms that cause impacts to propagate from the longwall mine panel to the swamps, which have been defined through the relationships described in the Bayesian belief network.

The mechanisms for impact can be categorised into three broad categories that reflect the time lag between mining and impact (Petts 1987):

- first-order impacts, which refer to the immediate impacts of subsidence (also called subsidence effects), such as cracking, shearing, tilting and reopening of bedding planes and joints within the sandstone
- second-order impacts, which refer to the impacts that result from subsidence effects, such as changes to swamp hydrology from altered groundwater or surface water flow paths, and water quality impacts
- third-order impacts, which are the result of changes to swamp hydrology and water quality, such as peat erosion and the ecological response of flora and fauna.

Third-order impacts can lag significantly from the first- and second-order impacts, with changes to swamp flora sometimes being identified decades after the longwall panel has passed through (NSW PAC 2009; and see Figure 7.1).



LW = longwall

Figure 7.1 Estimated timeline for impact, showing time lag between first-order impacts (subsidence effects), second-order impacts (hydrological impacts) and third-order impacts (ecological response). Timing for ecological impacts to occur could vary significantly and has not been studied in detail

A monitoring programme designed to establish baseline conditions aims to characterise the current condition of the swamps and establishes a baseline against which trends in condition, perhaps related to future impacts, can be compared. Conversely, a monitoring programme that aims to identify specific impacts early, so that management can be adapted, must focus on the first- or second-order impacts because these precede an ecological response.

The time lag associated with ecological impacts means that monitoring ecology is not an acceptable method for early identification of impacts. By the time an ecological impact is detected, subsidence effects, hydrological impacts and potentially peat destabilisation will have already occurred and it will be too late to mitigate impacts or to implement adaptive management to minimise impacts.

Furthermore, one of the main drivers of ecological response is change in hydrology and, specifically, a decrease in the groundwater level that reduces the duration of inundation of the swamp. Subsidence is not the only cause of a decline in groundwater level. Short-term climate variation (droughts and floods), and longer-term climate trends (climate change) are also drivers of change in groundwater level and, by association, inundation patterns in swamps. Hence, changes in ecological condition can be the result of several alternative ‘disturbances’. An effective monitoring programme design therefore needs to integrate subsidence, hydrological and ecological monitoring methods. While the remainder of this section focuses on ecological monitoring, it is strongly recommended that hydrological and subsidence monitoring methods are integrated with the approach discussed below.

The ecological monitoring approach recommended in this section aims to maximise the potential for impacts to be observed, and to be accurately attributed to longwall mining. As such, the approach incorporates a significant baseline monitoring programme that will help to fill one of the key knowledge gaps identified in this project: the ecological response to subsidence impacts. Understanding ecological responses to subsidence is necessary to design a monitoring programme that focuses on the most appropriate ecological parameters. This cannot be defined in a generic manner, and so this report recommends a process that can be followed to develop an effective monitoring programme. The approach involves significant effort in the initial characterisation of the system, to establish natural variability and further define monitoring density and frequency, and to reach agreement on the level of allowable impact.

7.2 The BACI approach

The before–after control–impact (BACI) approach (ANZECC & ARMCANZ 2000a, 2000b; Downes et al. 2002) for designing monitoring programmes involves data collection before and after impact occurs, from multiple sites within the expected area of impact (impact swamps) and outside it (control swamps). Data from control and impact sites allows the *difference* in a measured variable to be recorded at a point in time. The differences observed may be caused by subsidence or other natural variations between sites (sometimes called ‘nuisance’ differences). Combining this with observations at the control/impact swamps from before and after the impact allows a comparison of the *change* in the measured variable at each swamp over time. Assuming all other natural variables are equal between the swamps at each measurement date, the change observed between control and impact sites after impact can be attributed to subsidence.

A statistically powerful monitoring programme will enable even a small impact to be detected. This is considered desirable in the Sydney Basin, where ecological impacts can occur gradually over time and detection in the early stages of impact may allow management of further impacts. Ideally, the monitoring programme implemented for peat swamps would be sufficiently powerful to establish where there is no (detectable) impact with a high level of confidence (Faith et al. 1991).

The statistical power of a programme to detect impacts is improved by:

- increasing the numbers of sites sampled and adopting consistency in methods
- selecting measures/variables where differences are independent over time

- selecting control and impact sites that are similar before the disturbance
- selecting variables that are sensitive to the changes caused by the disturbance.

Statistical analysis of the monitoring data identifies significant departures from natural variation that can then be attributed to longwall mining.

Several important principles must be recognised in the monitoring programme design phase. The fundamental requirements of a good monitoring programme within the BACI framework were identified by Downes et al. (2002) and have been applied to swamp monitoring (Table 7.1). These principles are used in formulating the recommended monitoring and reporting approach in the following sections.

Table 7.1 Monitoring programme design principles

Requirements of a good monitoring programme (from Downes et al. 2002)	Principles for peat swamp monitoring
Effective monitoring requires understanding the nature, and temporal and spatial scales of both the disturbance event and the response.	Monitoring needs to detect 1) subsidence, 2) changes in hydrological regime and 3) ecological response. Subsidence impacts occur within a few weeks of mining, while ecological response may lag for several years after the initial subsidence impact. Swamps are discrete and many can occur above a proposed longwall. Longwall mining areas may be 6 km^2 (Dendrobium 3b), 20 km^2 (Metropolitan project), 200 km^2 (Bulli Seam Operations).
They key strategy for inference of impacts is to find some evidence for impact that cannot easily be explained away by various other processes, such as natural variation in the system. Support for an impact hypothesis is only found if the probability of that outcome is small, under normal circumstances, in the absence of impact. This pursuit of improbability provides a rationale for specific aspects of monitoring.	Changed hydrology is likely to lead to a change in ecological condition, but changed hydrology is not solely driven by subsidence. So, monitoring ecological condition alone is insufficient. Monitoring of direct subsidence and hydrological variables is also required. It would be ideal if existing studies could be used to clearly demonstrate that where subsidence has already occurred there has been a detrimental ecological response. If existing studies have not been completed then we recommend targeted surveys of impacted and unimpacted sites to demonstrate any existing differences in ecological condition that can be attributed to subsidence-related hydrological change.
Optimal design aspects include sampling control and impact locations—both before and after putative impact (so-called before–after control–impact (BACI) designs)—with, where possible, proper replication of each of these four elements. Replicated BACI-type designs allow, with relatively high confidence, separation of human-caused effects from natural processes.	Implementation of a multiple BACI (M-BACI) design is required when monitoring involves periods before and after potential impacts and at multiple sites where impacts are not likely to occur. This approach enables existing condition to be benchmarked and uses control sites to help determine if disturbances other than mining-induced subsidence (e.g. climate change) are also impacting on hydrological regime and ecological condition.
Variables chosen for monitoring should be efficacious: relevant to the questions asked, strongly associated with the putative impact, ecologically and /or socially significant and efficient to measure.	As described above, ecological variables are unlikely to meet the recommended criteria, especially with regards to being clearly linked to the putative impact. For this reason, first- and second-order variables need to be monitored as indicators of early impact and as surrogates for ecological response. However, it is also

Requirements of a good monitoring programme (from Downes et al. 2002)	Principles for peat swamp monitoring
	important that the relationship between mining → subsidence → hydrological change → ecological response be clearly demonstrated through targeted surveys of existing impact and control sites so that there is confidence in the use of first- and second-order variables as early indictors of likely ecological response.
The magnitude and form of unacceptable environmental changes (‘effect sizes’) should be negotiated and defined ahead of beginning a monitoring programme; it is impossible to prescribe universal effect sizes for biological variables.	Natural variability means identifying effects size is difficult in the absence of a long period of recorded data that enables the limits of natural variability to be defined. Where possible, historical data (e.g. from remote sensing) should be used to establish limits of natural variability and consensus reached on what then constitutes an unacceptable response (i.e. when the condition of the variable falls outside the limit of acceptable change). Monitoring is then aimed at detecting if and when an unacceptable response has occurred.
Monitoring programmes must be linked to management decision-making, such that particular triggers (e.g. an effect being detected) will result in some action being taken.	The monitoring programme needs to identify management actions that will be implemented if unacceptable responses occur in either first- , second- or third-order variables. Proponents must demonstrate that management actions will be effective at mitigating or minimising risk to ecological values.

7.3 Process for designing a monitoring programme

Recognising the BACI design principles and the need for an extensive baseline monitoring phase, the recommended monitoring approach incorporates three phases of monitoring, as shown in Figure 7.2.

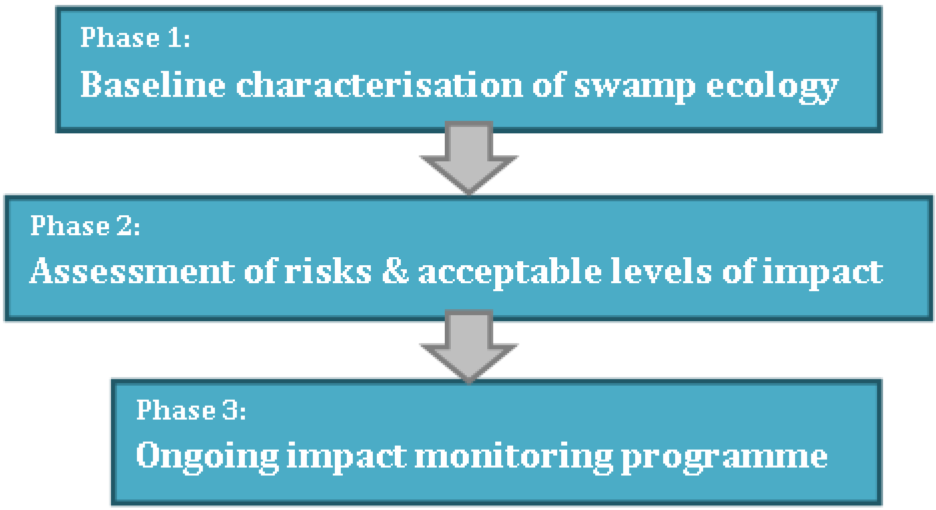


Figure 7.2 The three phases of an effective monitoring programme to detect potential impacts on peat swamps from longwall coalmining (see notes 1 and 2 at the end of the chapter).

The monitoring phases can be effectively considered as three separate programmes, with the results of each providing the information required to design the next phase. Each monitoring phase has different objectives and therefore there must be a separate study design process for each phase. The study design steps to be completed for each phase are consistent with ANZECC and ARMCANZ frameworks (2000b) and are shown in Figure 7.3. Although this monitoring framework focuses on ecological monitoring approaches, it could equally be applied to subsidence and hydrological monitoring programmes. Each component of the monitoring framework is explained in more detail in the following sections.

Under New South Wales legislation, mining proponents are required to prepare subsidence management plans (SMPs) for review and approval by an interagency government committee that predicts impacts and identifies how significant natural and built features are to be managed (NSW DP 2008). Swamps listed under the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act) classify as 'significant' swamps that must be considered within SMPs. Information from phases 1 and 2 would provide important data for the prediction of impacts and proposed management measures for swamps and need to be included in the SMP.

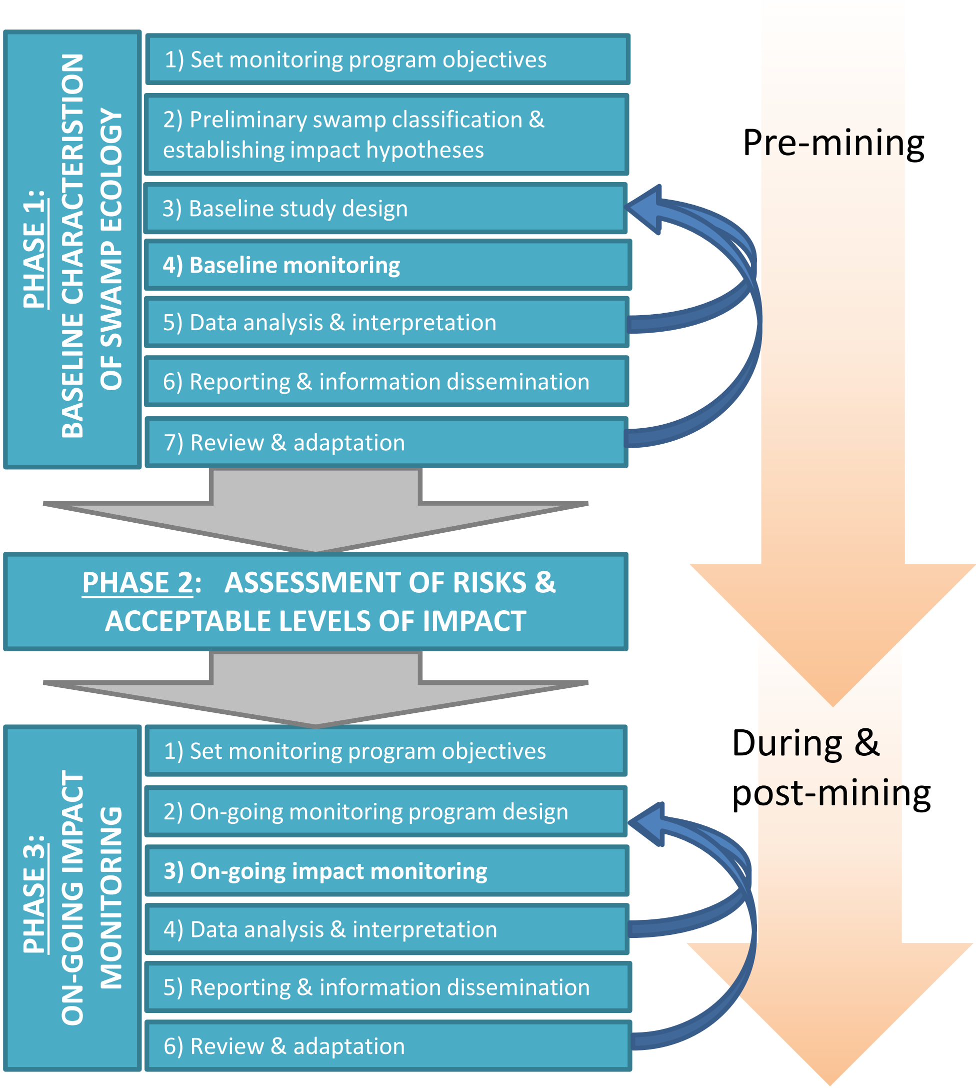


Figure 7.3 The steps at each monitoring phase for an effective monitoring programme.

7.4 Phase 1—baseline characterisation of swamp ecology

The information gained through the baseline assessment in phase 1 provides benchmark information against which future impacts can be assessed. Baseline monitoring involves an extensive field programme, which is required to design a statistically powerful ongoing monitoring strategy. The objectives of baseline monitoring are to:

- characterise current swamp ecology and condition. This data is critical for defining swamp value as part of the risk assessment and acceptable levels of impact (phase 2), and for selecting the most appropriate variables for monitoring
- establish the magnitude of natural variability in species and community composition at a spatial (between similar types of swamps) and temporal (e.g. seasonal) scale. This data is necessary for informing the selection of control and impact sites, and defining monitoring parameters and frequency for the ongoing monitoring of impacts (phase 3).

The specific steps involved in the baseline study are discussed in the following sections.

7.4.1 Setting objectives

The first step of any monitoring programme is to set monitoring objectives. For phase 1, the objectives should be to review, identify and characterise the natural features that require protection. Defining spatial and temporal variation is also a key objective of phase 1.

In further defining phase 1 monitoring objectives, the proponent may need to satisfy swamp-related requirements of:

- local, state and federal legislation and policy
- the Director General
- the community, including community values.

Information from these sources may provide additional detail for defining the objectives of baseline swamp characterisation, such as guidance on the level of detail required, the necessary outputs of the baseline characterisation or particular values that need to be preserved.

7.4.2 Preliminary swamp classification and establishing impact hypotheses

Before designing the field programme for the baseline characterisation, a preliminary classification of the swamps is required to establish an understanding of swamp occurrence, swamp characteristics and likely impact hypotheses. This knowledge is required to inform the specific study design requirements (see Section 7.4.3).

Initially, swamps within the Temperate Highland Peat Swamps on Sandstone Community (THPSS) must be identified. All swamps within the THPSS community can be considered as being of high value and must be protected from impact.

Swamps should also be classified into one of the three conceptual models: headwater swamps, valley infill swamps and hanging swamps, to give a qualitative indication of vulnerability. A first pass at this classification can be done by assessing swamp topography, since it is different for each type of swamp. Once classified, the conceptual models previously developed for each type of swamp indicate the potential for groundwater connection and the potential for subsidence impacts. Swamps in steeper topography (valley infill and hanging swamps) are more likely to be connected to groundwater, and are more prone to cracking and deformation of the underlying sandstone. These swamps are therefore more vulnerable to subsidence impacts than headwater swamps. An integrated monitoring programme would also require hydrological characterisation of the swamps, which would be useful detail for categorising swamps into conceptual model types. However, for an early classification, topography can be used to classify swamps.

To establish the likely impact hypotheses, an understanding of the likely areas and extent of impact is required. This will start to clarify the likely impacts to swamps in the vicinity of the longwall panel and help to define potential impact pathways. Using this information, the high-level hypothesis for impacts on the peat swamps can be defined and used to inform optimal variables for inclusion in the field-monitoring programme.

Swamps that are at higher risk of impact can also be determined. These include swamps at the edge of longwall panels (where differential subsidence causes more cracking and tilting of the sandstone) and in steeper terrain, such as valley infill swamps. These swamps should be prioritised for inclusion in the baseline characterisation monitoring phase.

7.4.3 Baseline study design

The recommended monitoring design framework for detecting impacts on peat swamps is a multiple BACI (M-BACI) design. This design samples multiple control sites and ideally multiple impact sites before and after a predicted disturbance, and aims to test whether relevant variables at the impact sites change relative to the state or condition of those same variables at the control sites. The use of multiple numbers of control and impact sites provides increased power or ability to detect changes because it allows the range of natural variability to be more confidently determined. The duration of the before and after impact sampling periods also has an effect on the ability to detect change, especially if the change does not occur in immediate response to the disturbance (i.e. there is lag response or a trend response rather than a step change).

The information required to enable an appropriate baseline study design to characterise the peat swamps is:

- preliminary understanding of swamp occurrence and broad characteristics
- preliminary understanding of area and extent of impact.

With knowledge of the likely area and extent of impact, and of the more vulnerable swamps, the specific components of the baseline characterisation can be determined. The design should consider (ANZECC & ARMCANZ 2000b):

- selection of sampling sites—all swamps inside the expected area of impact and a selection of swamps outside this zone should be included
- expected spatial variation—to ensure that variants are well represented in the site selection
- measurement parameters or variables—based on the impact hypotheses
- frequency of monitoring—based on the expected natural variation in measurement parameters
- precision and accuracy required—based on the impact hypotheses and measurement parameters
- cost-effectiveness—recognising the various methods available to efficiently measure the selected parameters to the defined level of accuracy.

Two years is considered to be the minimum period necessary for baseline monitoring (NSW DP 2008; NSW PAC 2009; NSW OEH 2012b). This is likely to be long enough to characterise the current spatial variation in swamp ecology. Ideally, the length of baseline monitoring would extend for as long as needed to establish the range of natural temporal variability in the swamp systems as well (i.e. taking into account longer-term climate variability). Remote sensing should be used to extend the historical data period up to 20 years into the past, to improve the understanding of temporal variation within and between swamps. The design of the ecological field programme needs to recognise that the data collected must also be suitable for calibration of remote-sensing data.

Baseline monitoring should be undertaken at all swamps that are within the predicted area of impact, as well as some outside the impact area that can be used as control sites (say an equal number as impact sites). There is no minimum number of control sites, although ideally, the more the better. The number of control sites needed depends on the degree of variation in selected variables; the greater the variability, the more sites needed to quantify variability.

Starting with a larger number of sites will allow optimum selection of swamps for inclusion in the ongoing monitoring programme, as effective ongoing monitoring requires either similar control/impact swamps, or a large number of control/impact swamps to be able to distinguish natural variation from subsidence impacts. Each of the initial swamp classifications should be represented in the baseline monitoring.

Within each swamp selected for survey, the number of subsample locations (replicates) will depend on the size of the swamp, and the variable being measured. A number of indicators for assessing ecological condition and health are available. The likely impact hypotheses defined in Section 7.4.2 identifies which variables are expected to respond to subsidence, and therefore which variables should be included in the monitoring programme. We recommend that a suite of indicators are measured for the baseline component of the monitoring programme (see Section 8). Throughout the baseline monitoring, indicators should be evaluated for their suitability to:

- help determine environmental condition
- be responsive to disturbance in a predictable way and ideally responsive to subsidence-induced disturbance.

Post-impact, the suite of monitoring indicators could be reduced to those that are known to respond to specific disturbances, as discussed in detail in Section 7.6.2.

The frequency of sampling each indictor depends on its response time step. For example, water levels would be monitored continuously because water level can change on a daily basis, whereas vegetation community structure may only need to be monitored once per year because it responds much more slowly.

7.4.4 Baseline monitoring

Ecological surveys should commence two years before any activity that may cause subsidence. Remote sensing can then be used to extend the historical data period up to 20 years into the past. The baseline characterisation should result in:

- documentation of flora and fauna species in the swamps, including identification of all threatened or vulnerable species and identification of invasive species
- detailed vegetation mapping of each swamp
- identification of the presence and abundance of threatened or vulnerable plants and animals in each swamp

- characterisation of swamp condition, including temporal variations in swamp extent and vegetation condition
- measurement of covariates/drivers of ecological response (e.g. hydrological regime).

At the end of the baseline monitoring programme, at least two years of field data should be available for calibration of remote-sensing data. The specific methods that are recommended to derive this information are discussed in detail in Section 8.

This baseline information can be used as inputs to a standard risk assessment framework that uses likelihood of impact and consequence of ecosystem loss to prioritise swamps for protection (phase 2), and hence the level of monitoring required.

7.4.5 Data analysis and interpretation

Once sufficient baseline data has been collected, variability can be determined and a suitable level of acceptable variance assigned. The baseline study should be used to compile existing data and collect new data at control and impact sites that can be used to benchmark differences between swamp types and determine acceptable levels of variability. These data can also be used to determine an acceptable limit of change (effect size) against which future impacts can be assessed (see phase 2 for advice on setting acceptable limits of change).

Establishing baseline variability of peat swamp ecology is the key to designing a pragmatic and achievable phase 3 ongoing monitoring programme. Understanding natural variability is important because swamp hydrology and ecological condition can also be impacted by natural and anthropogenic factors other than longwall mining. Natural factors include short- and long-term climate trends such as drought cycles and climate change. Anthropogenic factors include land use change, invasive species and human-induced fires. Monitoring has to be able to distinguish between natural variability and subsidence impacts. To do this, it is essential to establish the natural variations in ecological variables across peat swamp types before mining. Establishing the magnitude of natural variability also helps to identify acceptable limits of change (or effect size). Figure 7.4 shows an example of natural variability in an indicator measured over time, with occasional deviation outside the normal range of variability.

For most indicators of swamp condition there is unlikely to be sufficient data already available to define variability. However, remote sensing can extend the data available for defining temporal (and spatial) variation in some indicators. For example, satellite imagery may be used to assess vegetation greenness and evapotranspiration rates, which can be indicators of vegetation health. Remote-sensing imagery may be used to establish both spatial and temporal variability in some measures of vegetation condition.

Assessing spatial and temporal variability allows swamps that have similar ecological characteristics to be identified. This enables appropriate control and impact swamps to be selected for ongoing monitoring, as control/impact sites will ideally be similar in terms of ecological characteristics and temporal variations. This similarity increases the statistical power of the monitoring programme, allowing subsidence impacts of small magnitude to be identified and distinguished from natural variations.

The variability also determines how many sites should be monitored in the ongoing monitoring programme, as similar control/impact sites make identification of impacts easier, and therefore monitoring fewer sites could be justified. However, if swamps demonstrate high spatial and temporal variability, a large number of swamps must be included in the ongoing monitoring programme to increase the potential for detecting impacts.

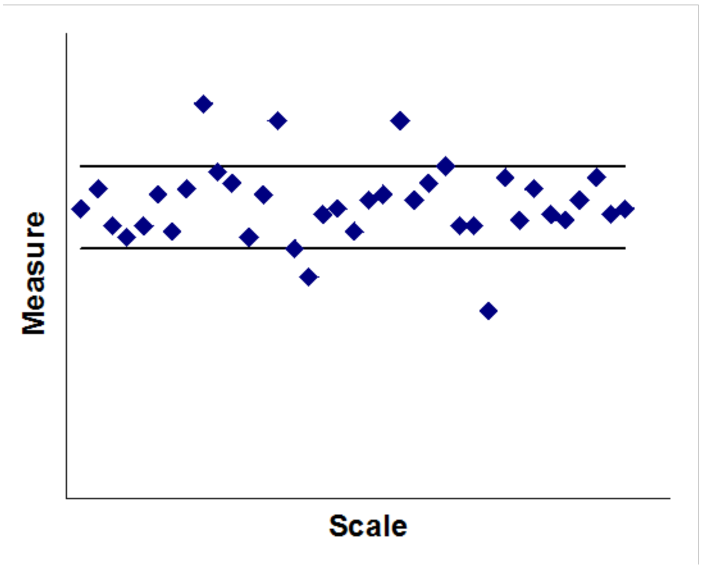


Figure 7.4 Example of natural variability in measured indicator with occasional deviation outside the normal expected range of natural variability.

7.4.6 Reporting and information dissemination

Reporting the data that characterises the swamps in the vicinity of the proposed longwalls enables proponents to engage stakeholders in the process. In particular, this information can form the basis of dialogue with regulators, and can inform the discussion and agreement on the acceptable levels of impact.

7.4.7 Review and adaptation

Any monitoring programme requires periodic review to assess the ongoing applicability of the decisions made at each step of this process. Priorities for ecological characterisation or variability analysis may change as more data on the swamps are collected. It is recommended that a review of the effectiveness of the monitoring programme is reported annually.

7.5 Phase 2—assessment of risks and acceptable levels of impact

7.5.1 Risk assessment

Risk assessments are used to prioritise monitoring and management activities through consideration of the consequences of an impact occurring combined with the likelihood that the impact will occur. Gaining an understanding of the ecological value of a swamp allows the consequence of impact on the swamps to be evaluated as part of a risk assessment. The preliminary classification (Section 7.4.2) has identified swamps that are part of the EPBC-listed THPSS community, and these can be considered as high-value ecosystems. Furthermore, baseline characterisation of the swamps (Section 7.4) identified the presence and abundance of threatened or vulnerable species, and swamp condition. These data allow an assessment of the consequence of impacts occurring in the swamps to be made.

The likelihood of each swamp being impacted by subsidence (cracking and tilting) is also required for the risk assessment. This should be based on predictive methods when mining plans are known, and involves review of subsidence and groundwater modelling predictions for each swamp. Guidelines on subsidence prediction are available in the report by Coffey Geotechnics (CoA 2014a).

The information gained through the risk assessment in this step allows swamps to be distinguished in terms of the likelihood and the consequence of impact. The ecological value of a swamp, the likely impacts of subsidence and the cost of conserving the swamp in terms of coal sterilisation or mitigation/remediation need to be considered.

7.5.2 Defining acceptable impact

Defining acceptable levels of impact for each swamp increases transparency in the management of environmental impacts and is critical for designing the phase 3 ongoing monitoring programme. Although the safest option is to state that negligible impacts should occur to the swamps (as recommended in NSW DP 2008; NSW PAC 2009, 2010), in reality some level of impact may be acceptable if future studies on ecological response indicates a level of resilience, or if the economic and social benefits exceed the environmental damage. The phase 1 baseline monitoring results and the risk assessment are key inputs into defining acceptable levels of impact.

Agreement on acceptable levels of impact, and the definition and variables that describe what is acceptable, determines the following components of the phase 3 ongoing monitoring programme:

- which variables need to be measured (i.e. the variables that are being used to measure acceptable impact levels)
- the frequency of monitoring for each variable (based on the response time of the variable)
- site selection (in terms of which swamps have been selected for preservation. Although, for the purpose of gaining more information on ecological response to subsidence impacts hydrological and ecological monitoring should be carried out on all swamps, even those that are not monitored need to be preserved).

Before mining begins, an acceptable level of impact (effect size) needs to be defined through workshops with all stakeholders.

One approach to defining acceptable levels of impact is to allow a defined level of impact to all swamps in the mining area. For example, it may be agreed that all swamps are of equal priority for preservation, and so impacts should be limited to a certain level for all swamps. The allowable level of impact needs to be defined in detail, and needs to be measureable. It may be a reduction in health or abundance of a sensitive species as an early indicator of ecological impact, loss of a certain sensitive species or change in vegetation distribution. The agreed impact therefore determines the variables selected for the ongoing monitoring programme.

Alternatively, the approach to defining level of impact may identify certain swamps for protection and others that are less important to preserve. This approach relies on classification, risk assessment and prioritisation of the swamps, and baseline analysis of variability, and must consider:

- value of the swamp—high-value swamps (such as THPSS) need to be preserved, while protection of lower value swamps may be less important
- level of risk—impacts to swamps in areas where subsidence impacts are expected to be severe may be sacrificed in favour of preserving other swamps

- spatial variability—if species composition is similar in all swamps, it may not be necessary to preserve all of them.

Where significant impact on swamps cannot be avoided, selection of offset sites may be an acceptable approach. This would also require evidence of spatial similarity between impact and offset swamps. In any case, monitoring of first- and second-order impacts is required to provide an early indication of the occurrence and potential severity of ecological impacts before they can occur. Adaptive management of mining can then be implemented before acceptable levels of ecological impact are exceeded.

7.6 Phase 3—ongoing impact monitoring

The objectives of an ongoing monitoring programme are to:

- identify when swamp health changes beyond the range of natural variability so that impacts can be attributed to longwall mining
- define ultimate impact on swamps.

Ongoing monitoring to detect impacts relies on the information collected in the baseline monitoring programme and the statistical analysis of baseline data to establish the range of natural variability. The understanding of natural variability and agreement on the level of acceptable impact are required as inputs to the ongoing monitoring programme to:

- identify which ecological variables will enable assessment against the agreed level of impact
- determine the appropriate frequency of ongoing monitoring for each variable
- select similar control and impact swamps so that subsidence impacts will be detectable
- identify similar control and impact sites for measurement of each variable.

The following sections discuss the recommendations for the ongoing monitoring programme.

7.6.1 Setting objectives

At a minimum, one of the objectives of phase 3 monitoring should be maintenance of swamp condition within the agreed acceptable levels of impact. The detailed knowledge of swamp characteristics and risks from phase 1 may also enable additional detailed objectives to be established. As for phase 1, objectives should also consider the requirements inherent in:

- local, state and federal legislation and policy
- Director-General requirements
- community requirements and values.

7.6.2 Ongoing monitoring programme design

A challenge with any monitoring programme is to know how many control sites are required. However, there is no minimum number of control (or impact) sites, although, ideally, the more the better. It is not possible to quantify the number or location of control sites as part of this study, because the selection of control sites needs to be based on the outcomes of phase 1. Monitoring frequency and density (number of swamps and different variables) can be reduced over the long term to those that are known to be responsive to impacts.

The number of control sites needed depends on the degree of variation in selected variables; the greater the variability, the more sites needed to quantify variability. However, there is also a trade-off with respect to variability across spatial scales. For example, to include a large number of control sites may require sites to be selected across a wide spatial scale, which may introduce additional variability. Furthermore, control sites and impact sites need to be as similar to each other as possible and behave similarly in the absence of the disturbance. To identify control sites, criteria need to be established to determine what qualifies as a control. Downes et al. (2002) outlines a process for selecting control sites that involves:

1. a review that identifies the natural sources of variability in the variables that will be used to monitor impacts, as discussed in Section 7.4.5
2. establishing criteria for site selection. For example, if discharge is identified as an important source of variability in the proposed monitoring variable (i.e. ecological response variable), then it will be important to match control sites for discharge regime
3. site inspections to identify sites with similar characteristics that match criteria
4. deciding on whether sufficient controls are available. This requires an understanding of the variability in the proposed monitoring variables and whether sufficient power is available to detect change. If too few sites are located, then criteria for site selection would need to be changed or relaxed to enable a larger number of sites to be included in the pool of potential control sites.

Control sites must be outside the anticipated zone of disturbance, as defined by subsidence modelling.

Impact sites include all swamps above the longwall panel and within the predicted area of subsidence. All impact sites should be monitored to establish baseline conditions before any potential impact.

Variable selection should be based on the level of acceptable impact and the agreed metrics for measuring these impacts. This is based on the historical range of natural variability (as determined through assessment of baseline ecological monitoring and remote-sensing data). Variables should be:

- able to determine environmental condition
- responsive to disturbance in a predictable way and ideally responsive to subsidence-induced disturbance
- repeatable and quantifiable where possible.

The frequency of monitoring depends on the response time of each ecological variable.

Detailed design of the monitoring programme also needs to incorporate management responses when certain trigger levels are reached. These trigger levels should be agreed as part of the discussion on the acceptable levels of impact. However, in most cases these trigger levels should relate to hydrological or subsidence impacts since these occur relatively quickly after longwall mining.

7.6.3 Ongoing impact monitoring

Ongoing ecological field monitoring should follow on immediately from the baseline monitoring. The duration of monitoring for each variable depends on the ecological response times and whether subsidence and hydrological impacts have ceased. If hydrological impacts have been observed, the ecological response may continue to occur for many years and, in this case, monitoring should continue until changes in ecology are no longer observed. Post-impact, the monitoring indicators could be reduced to those that are known to respond to specific disturbances.

7.6.4 Data analysis and interpretation

By establishing natural (or historical) variability in relevant indictors a range is created that describes the ‘normal’ function of peat swamps. When results are outside this range, the response may be attributed to some disturbance (e.g. subsidence impacts). Figure 7.5 shows an example of how monitoring that has established the relationship between, and variability in, a driver (hydrological) and ecological response variable can be used to determine if a disturbance (i.e. change in the driver variable) results in an effect beyond the natural range of variability at impact sites. Subsidence impacts are indicated by the migration of some variables outside the ‘normal’ range indicated by the black circle, into a new range indicated by the red circle. Assuming the ‘normal’ range was established over a period of time and is based on data that largely encapsulates natural variations, the migrating variables indicate that an external disturbance such as subsidence has caused the change.

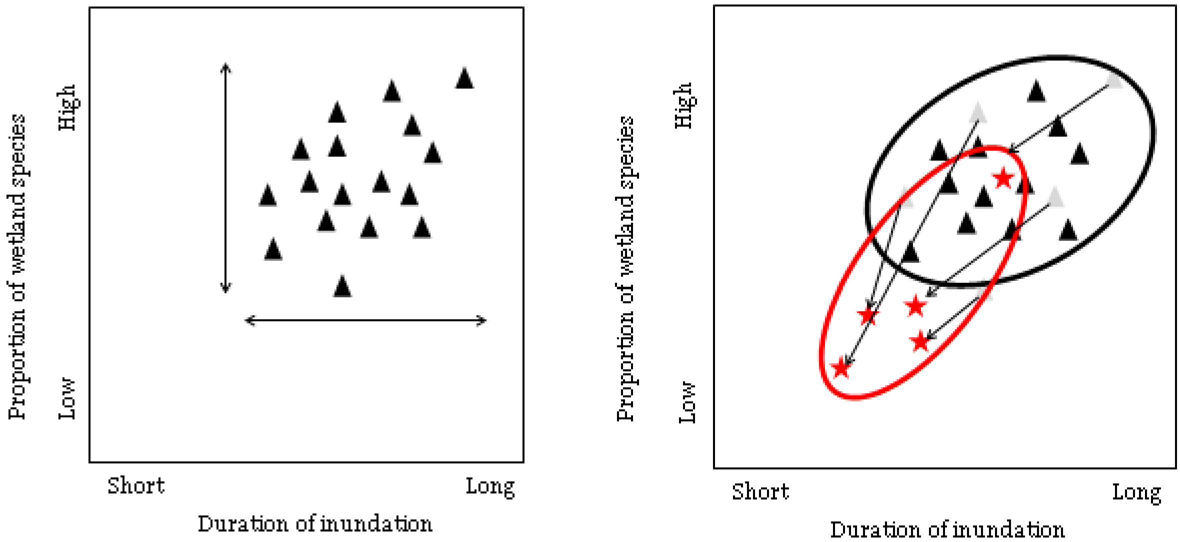


Figure 7.5 Conceptual diagram showing relationship between the variables driver (duration of inundation) and response (proportion of wetland species), the extent of natural variability and the response to a disturbance (change in duration of inundation).

Once variability has been determined and swamps have been monitored, the impact of subsidence can be detected using statistical tests. A range of statistical tests are used to compare data between control and impact and before and after time periods. Tests include analysis of variance (ANOVA), analysis of similarity (SIMPER) and t-test. The specific tests are determined by the monitoring design. M-BACI designs typically use ANOVA tests to compare data between sites and time periods.

An ANOVA compares the mean and variance between control and impact sites and between the before and after periods, and determines if there is a statistically significant difference in the mean after taking into account variability around the mean. A P-value is reported. A significant difference is usually confirmed if $P < 0.05$ (i.e. we are 95 per cent sure that a change has occurred). A $P < 0.1$ can be adopted if this has been agreed a priori; however, usually $P < 0.05$ is the adopted level of significance. The agreed level of impact (effect size) is also a measure of whether change has occurred (i.e. percentage of species lost).

Over time, trend analysis can also be applied, especially if changes to impact sites occur slowly over time (i.e. there is not a step change).

7.6.5 Reporting and information dissemination

Reporting of the data that describes the observed variations in swamp ecology and assesses whether this is due to longwall coalmining near the swamps is necessary so that compliance with agreed levels of impact can be reviewed. Reports should be written immediately if impacts are observed that are attributable to longwall coalmining, and should be publically available.

7.6.6 Review and adaptation

Any monitoring programme requires periodic review to assess the ongoing applicability of the decisions made at each step of the process. Priorities for preservation may change as more data on the swamps are collected, impact predictions may change and require a revised agreement on level of impact, or the detailed design components (site and variable selection, and monitoring frequency) may need to be adapted. A review of the effectiveness of the monitoring programme should be reported annually.

Peer review comments on Chapter 7

1. *Derek Eamus has commented in relation to Figure 7.3*:

   ‘that Phase One should contain baseline characteristics of swamp ecology, ecophysiology and hydrology (hydrological monitoring of the swamps is acknowledged in 7.3). The ecophysiology and hydrology components are missing and I would recommend their inclusion. By ecophysiology I mean swamp function, for example, ET (evapotranspiration) and surface temperatures. These attributes are likely to respond more rapidly than swamp ecology per se and therefore may allow a more rapid response to problems that arise from changes in local hydrology arising from longwall mining.’

2. *Ann Young has also commented on the importance of hydrological monitoring as a part of ecological monitoring—see comments at the end of Chapter 8.*

3. *Additional comment from Derek Eamus:*

   I note that in Chapter 7, there is much emphasis on the impacts of longwall mining on groundwater connectivity of the swamp per se. That is, the connectivity that is evident within the geographical extent of the swamp. However, further consideration of the potential for mining-induced changes in hydrology outside the swamps to impact the swamps is warranted. Changes outside of the swamp could change subsurface flows or run-on characteristics to the swamps.

8 Recommended monitoring methods

This chapter discusses the recommended monitoring methods to be used to establish a baseline of swamp ecology for swamps at risk of longwall mining subsidence and to detect impacts as they occur. The methods discussed relate predominantly to phase 1 of the monitoring programme, which should produce:

- documentation of flora and fauna species in the swamps, including identification of all threatened or vulnerable species and identification of invasive species
- detailed vegetation mapping of each swamp
- identification of the presence and abundance of threatened or vulnerable plants and animals in each swamp
- characterisation of swamp condition, including temporal variations in swamp extent and vegetation condition
- measurement of covariates/drivers of ecological response (i.e. hydrological regime; see note 1 at the end of the chapter).

Depending on the outcomes of the baseline monitoring and the analysis of natural variability, monitoring in phase 3 could be constrained to those variables that are known to respond to impact (if natural variability is small) or that may require additional monitoring methods if variability is large. Other methods are evaluated in Chapter 6. The results of the variability analysis also determine the site selection, variables to be monitored and monitoring frequency for the ongoing detection of impacts.

The ecological field survey programme should incorporate the following principles:

- Methods should be explicitly recorded and used consistently.
- Surveys should be undertaken by suitably qualified and experienced personnel.
- Survey locations should be recorded in sufficient detail and, where possible, marked on the ground to allow replication.
- Surveys should be quantitative where possible, since qualitative methods are subjective and can vary significantly through changes in monitoring personnel.

The following sections build on the evaluation of monitoring methods in Chapter 6. Specific methods and indicators considered most appropriate for peat swamp monitoring are recommended. For most of the monitoring outputs described below, the survey programme can be divided into a summer field programme and a winter field programme. The approach is based on ecological monitoring data collected over at least two years, and recommends using remote sensing to extend this baseline data collection period, to allow a longer period of variability to be used in the monitoring programme design.

8.1 Documenting and mapping flora species

Compilation of a species list can be used to determine the relative proportions of typical swamp species to non-swamp species. This is an important characteristic to measure because changes in the proportions of these species can indicate a shift away from typical swamp character towards a more terrestrial system that would occur as a result of altered swamp hydrology. Furthermore, threatened plant species are protected under the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act) and the New South Wales *Threatened Species Conservation Act 1995* (TSC Act). Activities likely to cause significant impacts on threatened species are not permitted without assessment and approval. Identification and monitoring of species protected under both Acts are necessary to enable effective planning of avoidance, mitigation and, potentially, offsetting under the relevant state and Commonwealth offset frameworks.

A flora species monitoring programme should be designed to answer the following questions:

- What are the flora species associated with each group of swamps (headwater swamps, valley infill swamps and hanging swamps)?
- Is the proportion of typical swamp species and non-swamp species changing?

This last question is answered by categorising flora species occurring in peat swamps as aquatic species, obligate wetland species, facultative wetland species and terrestrial species. The abundance of species in each category can be compared over time to show if there is a shift in the underlying vegetation community composition towards more terrestrial species. Categorising species as aquatic or terrestrial should be straightforward; these will be plants that only grow in water or plants that only grow in dry soil. Determining which plants are obligate wetland species (require saturated soil for survival) or facultative wetland species (thrive in saturated conditions but also survive long, dry periods) will need review of species physiology information and/or peer review by wetland plant specialists. Once a full list of plants compiled from field survey data has been categorised, a biannual species survey should occur each summer and winter. Seasonal surveys are necessary not only to ensure cryptic or ephemeral species are recorded, but also to enable seasonal variation in swamp character to be accounted for.

Survey plots used for field surveys should be:

- located to encompass all subhabitats present within a swamp
- marked with survey pegs or star pickets that are easy to find (painted with white or fluorescent survey paint)
- a standard size of 25 m^2.

All species encountered within the plots should be either identified in the field or sent to the Royal Botanic Gardens identification service, clearly marked with its location (unique plot identifier and geographic coordinates) and habitat information.

An estimate of cover–abundance for each species should also be recorded to capture quantitative information. A Braun-Blanquet cover–abundance scale is recommended, as this is a rapid way to collect quantitative information about each species. The method uses broad categories, resulting in a fairly coarse mapping scale, but has the advantage of reducing the influence of recorder subjectivity. This means there can be greater confidence in the consistency of results between recorders.

Targeted searches for threatened species should also be made concurrently as random meanders in suitable habitat. Suitable habitat can be identified using published threatened species habitat information available on the Species Profile and Threats Database[7] or the New South Wales Threatened Species website.[8] Random meanders should be at least 30 minutes long.

Where threatened species are identified, the location and extent of the population should be recorded (geographic coordinates). Population size can be recorded as a direct count for

[7] www.environment.gov.au/cgi-bin/sprat/public/sprat.pl
[8] www.environment.nsw.gov.au/threatenedspecies

small population sizes (between 1 and 100 individuals) or estimated for larger populations. There are a number of tested methods available for estimating plant population size. The simplest of these is to record frequency of plants in 1 m^2 plots and multiplying by the area inhabited by the plants; however, plants rarely exist in a consistent density across an area, and this simple method will give an underestimate or an overestimate. A time-efficient method for estimating plant density where plants are not distributed evenly is the 'nearest neighbour' method. A random point is located within the area the plant occurs, the plant nearest the random point is located and the distance from this plant to the next nearest individual is recorded. The nearest neighbour algorithm can then be used to calculate the density per area.

Targeted threatened species surveys in suitable habitat should continue for the duration of the survey programme, whether or not threatened species have been recorded in a previous survey. This is to offset the likelihood that threatened species will be overlooked in field surveys due to their frequently rare and patchy occurrence. Increasing the number of temporal replicate surveys will increase the likelihood of detecting threatened species.

A summary of the approach to mapping flora species is shown in Table 8.1.

Table 8.1 Summary of method for documenting swamp flora.

Method	Details
Survey method	Survey plots and species counts Random meanders
Required outputs	Complete species list with each species categorised as aquatic, obligate wetland, facultative wetland or terrestrial Cover–abundance scores for each species Threatened species locations and population estimates
Survey frequency	Biannually in summer and winter
Survey duration	Beginning at least 2 years before longwall mining, continuing until ecological equilibrium has been re-established

The monitoring period depends on the species present and their impact response times. If hydrological monitoring shows no change after a few years, it can be assumed that ecological impacts will not occur and the decision may be made to stop species monitoring. If hydrological impacts have been observed, the ecological response may continue for many years. Monitoring should continue until changes in ecology are no longer observed.

8.1.1 Invasive species

Extent of weed species is an indicator of overall swamp resilience to threatening processes. It is therefore important to monitor changes because resilience will be negatively impacted in the presence of other threatening processes, such as subsidence. Monitoring should be designed to determine the extent of invasive weeds and whether or not this extent is changing over time. Table 8.2 summarises the recommended methods for monitoring invasive weeds.

Although remote sensing can be used to detect invasive species, it requires a priori knowledge on the specific spectral and textural response to detect weed infestations. In most cases this is difficult to obtain and involves intensive field validation; field survey methods would therefore be more appropriate for monitoring this variable.

Field surveys should occur in 400 m^2 plots, marked by survey pegs or star pickets. From each plot the cover abundance of each weed species can be visually estimated using a Braun-Blanquet score or percentage or foliage projective cover. Smaller plots may be nested within the larger plot to increase the accuracy and ease of estimating abundance of small plant forms.

Monitoring should be undertaken at least biannually to account for seasonality of weed species. The number of sites needed will be adjusted in response to initial survey data, but, as a minimum, all swamps within 1 km of sources of disturbance (e.g. roads, farmland, urban development, extensive vegetation clearing) should be included in the initial surveys.

Table 8.2 Summary of recommended methods for identifying invasive species in peat swamps.

Method	Details
Survey method	Survey plots
Required outputs	Cover–abundance scores for each species
Survey frequency	Biannual
Survey duration	Beginning at least 2 years before longwall mining, continuing until ecological equilibrium has been re-established

8.2 Characterising vegetation pattern and distribution

Vegetation patterns correlate with the gradient of water availability and frequency of inundation. Changes to vegetation patterns therefore relate to changes in swamp hydrology.

Vegetation surveys should be designed to answer the following questions:

- Which vegetation communities are associated with each of the conceptual models?
- Which of these communities are water dependent?
- What is the extent of water-dependent vegetation?
- Is the extent of water-dependent vegetation changing?

A detailed map of vegetation communities should be compiled for each swamp. Boundaries of communities can be ascertained by field survey, using handheld GPS. Vegetation communities should be classified according to dominant species and structure. Table 8.3 summarises the methods recommended for characterising vegetation distribution.

Existing vegetation mapping data and vegetation classifications should be used to classify vegetation communities for each of the swamp conceptual models. A randomised plot design should be generated to adequately sample each of the conceptual models; this can then be refined based on access considerations and availability of existing available survey data. The number of plots will have to be sufficient to sample replicates of all vegetation communities present. The number of replicates will depend on variability so will need to be refined after initial baseline surveys are completed.

Survey plots should be marked on the ground with survey pegs or star pickets that are labelled with a unique identifier. A plot size between 25 m^2 and 100 m^2 should be selected and consistently used.

Minimum data collected from each plot should include a unique plot name, date, geographic coordinates, name of recorder, species present, stratum heights and species abundance in each stratum. For peat swamp vegetation surveys, either the foliage projective cover or Braun-Blanquet method of recording species abundance are suitable, but, whatever method is used, it should be used consistently to make sure data collected are compatible.

The plots should be surveyed at least annually, and surveys should take place during spring or summer, because fertile material helps when identifying a species.

Once preliminary information on vegetation distribution has been collected, it can be integrated with remote sensing to extend knowledge of vegetation distribution into the past, increase the spatial scale, and potentially decrease reliance on field methods in future monitoring. The historical data period available for this imagery is 10 to 15 years. The remote-sensing analysis of vegetation distribution would rely on the resolution and spectral bands available, with the imagery used for baseline definition. The remote-sensing monitoring programme suggested in this context is high-resolution satellite imagery, such as WorldView-2 or GeoEye, which would derive vegetation distribution based on 2 m resolution within the visible and near-infrared spectral ranges. Vegetation distribution can be mapped using a supervised classification technique, where training areas and its attribution, such as vegetation community classes, are derived from existing vegetation mapping such as 1:100 000 mapping, regional-based survey information or any available historical aerial photography.

Although the result of the vegetation distribution mapping when applied to multispectral imagery would not be able to distinguish individual species, it will be able to identify vegetation communities with similar appearance/textural and similar spectral response. For example, it is unlikely that sedgelands and specific grasses would be differentiated; however, grassland communities could be grouped and delineated. The supervised classification methodology can be applied to a range of multispectral imagery, for example 2 m WorldView-2, where the pixel size, or the ground sample distance, would define the level of detail delineated within the swamp.

Characterising vegetation pattern and distribution using high-resolution data has historically been completed with natural colour aerial photography. However, its use in a monitoring context is not widely adopted, especially for peat swamp monitoring. This is because of the limitations of the spectral response in a natural colour image and the subsequent analysis being reliant on manual delineation of vegetation communities. High-resolution multispectral data, however, removes the reliance on manual delineation of vegetation extent because it records a wider spectral response in the visible and near-infrared bands. Despite the high acquisition cost, the level of detail captured using multispectral high-resolution satellite data will be a significant added value for monitoring activities; these bands are used to differentiate vegetation from its surrounds and distinguish vegetation communities.

Table 8.3 Summary of methods for characterising vegetation distribution.

Method	Details
Survey method	Survey plots and species counts Remote-sensing analysis using a supervised classification technique
Required outputs	Vegetation community maps for each swamp
Survey frequency	Annually in spring or summer
Survey duration	Beginning at least 2 years before longwall mining, continuing until ecological equilibrium has been re-established

8.3 Characterising swamp condition

8.3.1 Determine variations in swamp extent

The first step in establishing baseline condition from remote sensing is to determine historical variations in the extent of the peat swamp. A combination of analyses would help to establish natural variations in peat swamp ecology, including:

- image analysis using the normalised difference water index (NDWI) to detect open water and, to some extent, water under the canopy. This technique would need to be applied in both wet and dry periods, to provide information on the maximum and minimum fluctuations of open water in the swamp
- image analysis using the normalised difference moisture index (NDMI) to detect areas with higher moisture content. This analysis assumes that the peat swamp would have higher water or moisture content either from water, wet soil or vegetation. Similar to the water index, applying this technique in seasonally wet and dry periods would provide an indication on the maximum and minimum extent of the swamp. This method would complement the water index analysis, where non-waterlogged areas of the swamp could be identified
- image analysis using normalised difference vegetation index (NDVI) to inform the vegetation extent. Although this method could not be used to generate information on specific vegetation communities in the swamps, it could be used to distinguish vegetated and non-vegetated parts of the swamps.

These methods would be applicable to multispectral datasets such as WorldView-2 that have spectral responses in the red, green, near-infrared and shortwave infrared bands. The result of the image analysis would require domain knowledge from an ecologist to refine the swamp extent. For example, knowledge on the seasonal condition of the swamp during dry months, especially on the presence of open water and moisture, will help refine areas that need to be included or excluded from the swamp extent. Similarly, other supporting information such as topographic information (e.g. elevation between 600 and 1200 m) would be useful for refining boundary extent by isolating likely locations where temperate highland peat swamps occur and in determining swamp type. Archive data is available from the early 2000s until present; however, because these commercial satellites only capture imagery on request and do not routinely monitor the surface, specific dates may be difficult to obtain.

The use of remote-sensing technology in defining baseline extent for hanging swamps would be limited, even with high-resolution imagery such as WorldView-2 or GeoEye because existing remote-sensing technology is dependent on nadir downward-looking perspectives. This makes it difficult to detect swamps on slopes, particularly on steep slopes where shadowing contributes to obscuring the swamps, making them more difficult to detect. Remote-sensing methods are therefore most applicable for valley swamps and headwater swamps. The methods for determining variations in swamp extent are summarised in Table 8.4.

Table 8.4 Summary of survey methods for characterising variations in swamp extent.

Method	Details
Survey method	Normalised difference vegetation index analysis of WorldView-2 or GeoEye Field GPS readings of swamp edges
Required outputs	Range of swamp extents over the past 15 years (or more)
Survey frequency	Biannually; in late summer when available water is at a minimum and in late winter when vegetation response to water availability is at a peak (see note 1 at the end of the chapter)
Survey duration	At least 10 years before longwall mining, continuing until hydrological and ecological equilibrium has been re-established

Remote-sensing data should be verified in the field. This can occur concurrently with other field surveys, and should involve collection of GPS coordinates at swamp boundaries. The data can then be used to verify the remote-sensing interpretations and used as an input to refine boundary definition.

8.3.2 Vegetation condition

Vegetation condition is closely related to water availability and peat stability, so this is a fairly direct indicator of changes in swamp hydrology. Survey and monitoring of vegetation condition should be designed to answer the question ‘what is the health of the swamp vegetation for each of the conceptual models, and is it changing over time?’ Variables to monitor are extent of vegetation (live and dead), bare ground and peat cracking.

The most suitable methods for monitoring vegetation condition in peat swamps are airborne and satellite-based remote-sensing techniques, as field surveys on vegetation condition are prone to subjectivity. Multispectral indices such as the NDVI and the NDMI can be used to analyse remote-sensing imagery. Benefits of using remote-sensing methods are as follows:

- Historical data can be used to gain access to true ‘before impact’ data and to establish variability of vegetation health before mining impacts.
- The cost of methods may be lower that field data collection.
- Weather and access constraints are minimised.

WorldView-2 is preferred to GeoEye as the imagery source because of its greater spectral range, allowing more information to be derived about vegetation condition. Archive data is available from the early 2000s but specific dates may not be available as the commercial satellites do not routinely capture imagery.

Vegetation condition can be determined through vegetation density from supervised classification and from NDVI, which measures vegetation vigour, or greenness. Patterns of greenness response over time are indicative of different growth rates and phenology, and seasonal wet and dry season analysis over a period of time will determine natural fluctuations in variability. NDWI would provide additional information on certain vegetation communities that have high moisture content, such as sphagnum.

The information from the image analysis should be correlated with supplementary data such as climatic and extreme condition events, where periods of intense rainfall and drought or fire may influence vegetation condition and be excluded from the thresholds of natural variability.

Evapotranspiration (ET) rate can be observed using remotely sensed data and, as it provides a measure of water availability, can be used to establish rates of vegetation condition. ET has been used extensively to map groundwater-dependent ecosystems. Since swamps are often groundwater dependent, correlations can be made between ET and vegetation health. Establishing baseline conditions for ET would involve analysing mean monthly summer ET because groundwater-dependent ecosystems are more apparent in the landscape during summer when rainfall is limited and after rainfall in winter. This would provide a value of natural variation in ET.

Anomalies on the remote-sensing analysis would need to be verified with ground truthing. For example, fire events or vegetation disease could cause drastic changes in the condition monitored by the remote sensing and present anomalies or false positives.

The most frequent monitoring interval using remote sensing from Landsat would be bimonthly. Change of swamp extent and vegetation distribution could be detected by using the image analysis techniques described above and visual inspection of the changing boundaries. However, it is unlikely that such a frequent interval would be necessary to detect ecological change. It may be more appropriate to monitor every season for the first 5 years and then select a less frequent monitoring period for subsequent analysis, say seasonal wet and dry for every 1 or 2 years for at least 10 years.

The NDVI could be derived from image data taken at regular intervals, and the NDVI short-term trend for vegetation community groups could be used to infer vegetation condition in the swamp. Accelerated changes outside its natural pattern may indicate impacts in vegetation condition due to mining; however, this would need to be validated with a localised dataset such as climatic and rainfall data. Similarly, a declining trend of the NDVI may indicate that vegetation condition has declined. All these methods would need to be supplemented by field verification to validate and refine the delineation of extents. In addition, data collected from natural colour aerial photography can be used to visually confirm the result.

As discussed in Section 6.2.1, the enhanced vegetation index (EVI) would better provision for data errors due to atmospheric and soil influences than the NDVI. However, because the EVI has been recently developed, it had not, at the time of writing this report, been sufficiently tried in similar environments to those of the peat swamps. Therefore, the NDVI, where benefits and limitations have been thoroughly studied, is recommended for use in peat swamps.

Ultimate impact may be manifested in the presence of a greater proportion of bare soil or evidence of gully erosion. Land cover mapping during a dry period at an interval of 5 years and 10 years would be appropriate to detect any ultimate impact. Identifying gully erosion would involve verification with a stream network dataset and, ideally, a high-resolution elevation model that would be able to determine bare ground situated on downward gradients. Field survey would be needed to verify and refine the extent. The approach to determining variations in swamp extent are summarised in Table 8.5.

Table 8.5 Summary of methods for mapping vegetation condition.

Method	Details
Survey method	Normalised difference vegetation index analysis or unsupervised classification of WorldView-2or GeoEye imagery Field verification and visual inspection of natural colour aerial photography
Required outputs	Vegetation condition mapping
Survey frequency	Seasonally (four times a year) for 5 years, then summer/winter for at least 10 years
Survey duration	At least 10 years before longwall mining, continuing until hydrological and ecological equilibrium has been re-established

Note: See note 2 at the end of the chapter.

8.4 Fauna species

Swamp fauna dependent on water include wetland frogs and birds, wetland-dependent invertebrates and reptiles. Monitoring these groups gives a measure of conservation significance, as well as an indication of changing hydrology. Several threatened fauna species dependent on temperate peat swamps are protected under the EPBC Act and the TSC Act. Activities likely to cause significant impacts to threatened species are not permitted without assessment and approval. Identification and monitoring of species protected under the Acts are necessary to enable assessment of impacts and effective planning of avoidance, mitigation and, potentially, offsetting under the relevant state and Commonwealth offset frameworks.

The fauna monitoring programme should be designed to answer the following questions:

- What swamp-dependent species are present? How does diversity or population size vary? Is the diversity or population size of wetland-dependent species declining?
- What other threatened species are present? How does diversity or population size vary? Is the diversity or population size of threatened species declining? Is the group of swamps core habitat for a threatened species population?

Surveys for swamp-dependent fauna should include wetland frog surveys, wetland bird surveys and targeted surveys for threatened species. Species targeted for survey will be selected from the threatened ecological community listing advice, any relevant previous ecology surveys in the locality and from database searches (EPBC Protected Matters and NSW DEH Wildlife Atlas). The recommended monitoring methods are described in the following sections and summarised in Table 8.6. Remote sensing cannot be used for fauna surveys.

8.4.1 Wetland frog surveys

Monitoring of wetland frogs is necessary to establish the baseline ecological character of each group of swamps. Suitable methods for detecting wetland frogs in peat swamps—visual encounter surveys, audio transect survey, static call surveys and automated call recording—were described in Section 6.1.2.1 and are detailed further in DEWHA (2010a). Frog surveys should be undertaken only in optimal weather conditions (warm, low wind velocity, high humidity, post-rain events) and incorporate all of these methods to maximise the likelihood of detecting all species present. Surveys should include nocturnal and diurnal intervals during times of peak activity.

If conducting targeted surveys for giant burrowing frog, systematic nocturnal and diurnal searches should be done in suitable habitat by two observers in 100 m × 50 m transects over 30 minutes (Queensland EPA 2005). In addition to visual encounter surveys, an audio transect survey should be undertaken. As detection accuracy improves with number of successive visits, at least four surveys should be undertaken at each swamp in each breeding season (autumn) following heavy rainfall. An automated call recorder can be set-up at the time of the first survey and retrieved after the fourth survey.

8.4.2 Wetland bird surveys

Monitoring of wetland birds is necessary to establish the baseline ecological character of each group of swamps. Suitable methods for detecting wetland birds in peat swamps—diurnal surveys, call broadcast surveys and nest counts—were described in Section 6.1.2.3. A combination of methods should be used to maximise species detected and timed to coincide with the arrival or departure of migratory species.

As repeated sampling over multiple days and at different times of the day improves detection rates, there should be a minimum of four survey days at each swamp twice a year.

8.4.3 Wetland invertebrate surveys

Monitoring of invertebrate species known to be sensitive to water quality changes, such as some dragonflies (e.g. giant dragonfly) and damselflies, may be a suitable indicator of peat swamp condition. Therefore, targeted searches for the threatened giant dragonfly should be included in an invertebrate survey programme. Specific survey techniques for giant dragonfly are diurnal searches for adults and exuviae using handheld sweep nets along 100 m transects in suitable habitat.

Sites with known populations of this species should be used as reference sites to indicate when adults emerge, and hence when other sites should be surveyed. Surveys should occur annually, with weekly visits to each swamp throughout the adult lifecycle (November, December and January). Ideally, surveys should be between 10 am and 3 pm on days with less than 50 per cent cloud cover and low wind velocity.

Macroinvertebrate surveys can also be used to measure the ecological response to changes in habitat and water quality in areas where waste water discharge or other water quality changes may occur. Since rapid bioassessment techniques are applicable to instream and wetland habitats rather than swamps, the samples would need to be taken from the surface water flowing into the swamps and water flowing out of the swamps. The metrics used to indicate macroinvertebrate health (as a response to changes in habitat and water quality) include AusRivAS (Turak & Waddell 2002), SIGNAL (Chessman 2003) and POET. Any changes observed between the baseline and the ongoing impact monitoring phases could be used to suggest water quality impacts.

8.4.4 Wetland reptile survey

Field survey methods for wetland reptile species suitable for peat swamps are diurnal hand and visual searches, in which a defined area of suitable habitat is searched for a defined period of time, as summarised in Table 8.6. Searches should occur between mid-morning and late afternoon, but this may vary according to local weather conditions. As detection accuracy improves with the number of successive visits, at least four surveys at each swamp should be undertaken each year in the summer months when the species is likely to be active. Care should be taken to minimise destruction of habitat while conducting hand searches.

Table 8.6 Summary of fauna survey methods suitable for peat swamp monitoring.

Method	Details
Survey method	Frogs: Visual encounter surveys, audio strip transects, static call surveys and automated call recording Birds: Diurnal surveys, call broadcast surveys and nest counts Invertebrates: Diurnal searches for adults and exuviae Reptiles: Diurnal hand searches and diurnal visual searches
Required outputs	List of fauna species present
Survey frequency	Annual for all faunal groups, except birds which should be surveyed biannually
Survey duration	Frogs: At least four surveys per breeding season Birds: A minimum of four survey days at each swamp twice a year Invertebrates: One day per week throughout November, December and January Reptiles: At least four surveys at each swamp each year

8.5 Summary of recommended monitoring approach

By the time an ecological impact is detected, subsidence effects, hydrological impacts and, potentially, peat destabilisation will have already occurred, and it will be too late to mitigate impacts or to implement adaptive management to minimise impacts to the swamps. A fundamental principle of monitoring ecological impacts of subsidence is therefore:

Recommendation 1: For an early indication of potential ecological impacts, ecological monitoring should be integrated with monitoring of subsidence effects and hydrological impacts.

Information linking subsidence effects to ecological impacts is limited, with little information that specifically describes how swamp ecology responds to changes in the surrounding environment. There is also very little understanding of the natural variations in swamp ecology over time. These knowledge gaps mean that current monitoring programmes are not designed to measure specific ecological changes that are known to occur in response to subsidence. Because of this, monitoring is usually unable to distinguish between changes due to natural ecological variation and changes caused by subsidence.

The limited knowledge of swamp variability and ecological responses to subsidence indicates that an appropriate basis for designing an ecological monitoring programme is:

Recommendation 2: Adoption of a multiple before–after control–impact (M-BACI) approach to monitoring swamp ecology.

A monitoring approach has been developed (see Section 7) to maximise the potential for impacts to be observed, and to accurately attribute them to longwall mining rather than to natural variations like drought, seasonal variations or fire. As such, the recommended approach incorporates a significant baseline monitoring programme that aims to fill one of the key knowledge gaps identified in this project: the ecological response to subsidence impacts. This cannot be defined in a generic manner, so a key requirement of an ecological monitoring programme is:

Recommendation 3: Monitoring must include an extensive baseline monitoring programme that establishes natural variability so that natural variations in ecology can be distinguished from variation caused by subsidence impacts.

A monitoring programme that is capable of detecting impacts on the swamps and attributing the impacts to longwall coalmining rather than to natural variation must incorporate three phases of monitoring:

Recommendation 4: Consideration of the following steps is required to develop an effective monitoring programme.

Phase 1: Baseline characterisation of swamp ecology

▼

Phase 2: Assessment of risks & acceptable levels of impact

▼

Phase 3: on-going impact monitoring program

The outcome of each phase of monitoring informs the design of the subsequent monitoring phase.

A range of monitoring techniques can be adopted for the baseline monitoring in phase 1, but the methods selected must deliver:

- documentation of flora and fauna species in the swamps, including identification of all threatened or vulnerable species and invasive species
- detailed vegetation mapping of each swamp
- identification of the presence and abundance of threatened or vulnerable plants and animals in each swamp
- characterisation of swamp condition, including temporal variations in swamp extent and vegetation condition
- measurement of covariates/drivers of ecological response (i.e. hydrological regime).

The specific parameters included in the baseline monitoring phase should be informed by a preliminary swamp classification that includes the development of hypotheses describing potential impacts on the swamp ecology, and therefore identifies the ecological parameters that are expected to respond to longwall mining impacts. These are the parameters that the baseline monitoring should be built on.

Baseline monitoring needs to continue as long as necessary to establish the range of natural ecological variability. Because field surveys are resource intensive, remote sensing should be used to extend the historical data record. Recommendations regarding the duration of baseline monitoring are therefore:

Recommendation 5: Field surveys for baseline monitoring (phase 1) should begin at least two years before longwall mining, and remote-sensing data should be reviewed as long as possible into the past.

When baseline monitoring has characterised the ecology of the swamps and provided information on the magnitude of natural variability in ecology, health and composition (phase 1), an informed risk assessment that helps define the acceptable levels of impact can be done (phase 2). The outcomes of phases 1 and 2 directly control the design of the monitoring programme for phase 3:

Recommendation 6: The parameters included in the ongoing impact monitoring programme (phase 3) should be those that were observed to be responsive to change and for which natural variability was well defined by baseline monitoring.

The duration of ongoing impact monitoring depends on the species present and their impact response times. If hydrological impacts have been observed, the ecological response may continue to progress for many years.

Recommendation 7: Ongoing impact monitoring should occur until both hydrological and ecological monitoring indicates that the system is stable.

Peer review comments on Chapter 8

1. NSW Government agencies comment: The timing of surveying should not assume water availability is less in summer and more in winter—timing needs to be based upon rainfall.
2. Ann Young suggests the following hydrological monitoring to help understand changes in flora and fauna:

   Rainfall, evapotranspiration and run-off measurement or reliable estimation

   - watertable depth and fluctuations in response to rainfall
   - water quality especially pH, DO, dissolved Fe
   - flow or chemical analyses that allow estimation of groundwater contribution to swamp hydrology
   - swamp sediment and peat characteristics, especially peat type, humification and distribution; sediment texture (sand/clay percentages), organic matter content, moisture content. There is some evidence that swamp sediments retain soil moisture for some time after the watertable drops after subsidence, and this may correlate with rate of ecological change.

   In addition it is suggested that monitoring methods may need to be tailored for individual swamps and targeted monitoring regimes could be developed to answer such questions as:

   - how important is that swamp's vegetation and faunal assemblage regionally and as a representative of the THPSS EEC?
   - how important is the hydrology of that swamp to the environmental flows downstream and the ecological health of the subcatchment containing the swamp?
   - what are the likely direct impacts of mining-related subsidence, such as bedrock cracking, change of slope, rockfalls, etc. on the swamp?
   - what are the secondary impacts, such as change in swamp hydrology and longer-term changes in ecological function?
   - how are these impacts to be measured and what changes will be considered serious enough to warrant intervention such as a change to an approved mining plan?
3. Derek Eamus comments: One particular aspect that perhaps could warrant greater attention is the use of remote sensing to monitor changes in swamp function, rather than just spatial extent or

structure (although these are important too). As noted in the report, changes in swamp structure are likely to be relatively slow and may occur after significant changes in hydrology/subsidence have occurred. In contrast, changes in swamp function may be detectable earlier. Examples of function that can be determined with remote sensing include rates of evapotranspiration (ET), surface temperature and productivity. By measuring such functional attributes of swamps potentially affected by mining and (a) adjacent native woodlands and (b) 'control swamps' located in areas without threat from mining, simultaneously it is likely that the effect of natural changes in climate (rainfall, vapour pressure deficit, drought) on the signal obtained from the swamp of interest can be removed (by taking the ratio of vegetation functional signals for the swamp of interest and control swamps/surrounding native vegetation. However, I do note that groundwater-dependent vegetation (the swamps), is likely to show a different response to drought, for example, compared to the surrounding native vegetation that is not accessing groundwater, hence the need for control swamps.

Professor Eamus has also suggested the report would benefit by adding citations to references that describe the application of the proposed monitoring methods more fully. Some suggested references are provided in the References, under 'Additional references provided by peer reviewers'.

9 Knowledge gaps

This study has involved a literature review and sensitivity analysis, and has developed a monitoring strategy for establishing a baseline and detecting impacts on the peat swamps. Several knowledge gaps have been identified, as well as opportunities for potentially streamlining the assessment process for developing new mines. This section describes each knowledge gap and presents a high-level research proposal aimed at addressing them.

9.1 Researching natural variability and ecological response to disturbance

The most fundamental knowledge gap is the lack of knowledge of the ecological response to subsidence impacts. Although it is broadly acknowledged that there is a significant ecological response over a relatively long timeframe, there are few data on the sensitivity of specific species to changes caused by subsidence (aside from what can be inferred from knowledge of the habitat requirements of certain species). Measured data relating to species response to disturbance is required to enable detailed design of a monitoring programme that will be able to detect ecological impacts and attribute them to subsidence caused by longwall mining. This requires an understanding of the natural variability of swamp ecology, which has been recommended as the initial step of baseline monitoring described in this report (Section 7.4).

Although establishing the range of natural variability has been recommended as part of a proponent's monitoring plan, a pilot research project that compares ecological variation between impacted swamps and non-impacted swamps would provide valuable information to help understanding of the spatial variability of the swamps. This programme could also:

- identify species that are most sensitive to impact (i.e. they occur commonly in non-impacted swamps and rarely in impacted swamps), which may suggest a suite of useful indicator species for monitoring
- identify the variables that are responsive to subsidence impacts and are therefore useful for ongoing monitoring.

Such a research programme would ideally cover as large an area as possible, and should include swamps from each geographic region in the Sydney Basin, including the Southern Highlands, Woronora Plateau, Blue Mountains and Newnes Plateau. It should also include swamps from each conceptual model type (i.e. headwater, valley infill and hanging swamps). The specific tasks would be to undertake:

- detailed vegetation mapping of each swamp selected for study (impacted and non-impacted)
- fauna surveys
- species abundance surveys.

To address the key knowledge gap, this programme must establish ecological variability between impacted and non-impacted sites. To do this, it must consider the first- and second-order impacts of subsidence (rock deformation and hydrological impacts). Measurements of all three levels of impact (geological, hydrological and ecological) need to coincide so that changes in rock integrity and hydrology can be linked to ecological response. This knowledge would enable development of a more effective and efficient swamp monitoring programme that is more likely to enable early identification of impacts.

The programme would also benefit from the addition of temporal data to the understanding of spatial variability. Remote sensing (with field verification) could extend the interpretations of swamp variability into the past, allowing an assessment of variability over time, including an assessment of ecological response in impacted sites and natural ecological variability in non-impacted sites. Data could be used to assess three variables over time for the selected swamps:

- changes in swamp extent
- changes in vegetation distribution within the swamp
- changes in vegetation health.

This programme would provide data to begin addressing the fundamental knowledge gap in the protection of peat swamps, which is how swamp ecology responds to disturbance. It would allow design of more effective monitoring programmes, and information that could be used to develop adaptive management or remediation strategies for the swamps.

The data gained from the programme recommended above could also be used to refine the probabilities in the Bayesian belief network (BBN). Often the probabilities used in a BBN can be based on previous studies that have quantified the probability of impacts occurring. However, in this case, the probabilities were based solely on specialist knowledge of the peat swamp's function. The specialists' confidence in assigning probabilities to the BBN varied for different nodes. For example, the probability of rock deformation occurring in response to certain mining parameters is relatively well established. Data also qualitatively supports the assumption that changes in groundwater levels and therefore swamp inundation will occur in response to subsidence effects. However, there is little information (no quantitative and little qualitative information) that describes the ecological response to subsidence impacts. Adding quantified data to the BBN would strengthen its potential to be used as a risk assessment tool, making it more useful for predicting impacts on ecology for individual swamps in the early stages of the regulatory approvals process. Depending on the confidence in the quantified data, it may also help define trigger levels for ecological impact in terms of inundation thresholds required to maintain particular species, and potentially the ecological community overall.

9.2 Validation of remote-sensing approach for monitoring peat swamps

One of the key recommendations in this report for monitoring is the use of remote-sensing data to reduce reliance on field surveys. Remote-sensing data could be used to give frequent and historical information on vegetation health and community patterns, which are useful variables when assessing impacts. This information can be used to assess where impacts may have occurred in the past and where continued monitoring can identify future ecological impacts. Any remotely sensed data should be verified if it is to be used to draw conclusions at a swamp scale. This would involve assessing remote-sensing images and field verification of the interpretations.

A more sophisticated analysis method called SEBAL gives actual measurements of evapotranspiration (ET) (e.g. in mm/month). This data would provide the quantitative information that is currently lacking in the BBN (assuming that ET is a proxy for vegetation health), and would enable spatial and temporal changes in ET to be linked to subsidence effects. This method is recommended for maximum detail and accuracy; however, it is still relatively expensive compared to the normalised difference vegetation index and normalised difference moisture index techniques. SEBAL would also allow a stronger comparison of impacted and non-impacted swamps and allow detection of changes over time.

9.3 Integration of advice for designing a monitoring programme

This report focuses primarily on the approach to ecological monitoring. However, it must be recognised that the ecological response to subsidence and changing hydrology lags significantly behind longwall mining. As such, it is critical that the overall monitoring strategy integrates subsidence monitoring and hydrological monitoring. There are two reasons for this:

- To identify impacts before the ecological response occurs. If subsidence effects or impacts to hydrology are identified, there may be sufficient time to change mining parameters so that impacts to ecology are minimised.
- To enable the ecological response to subsidence and hydrology impacts to be defined.

10 Summary of key findings

10.1 Conceptual models

An accurate conceptualisation of the environmental relationships that control the presence of the peat swamps is required to assess sensitivity of the swamps to change and to recommend the most appropriate monitoring regimes. Three conceptual models describe the peat swamps:

- headwater swamps—formed near catchment divides where topographic gradients are shallow. These swamps are predominantly reliant on rainfall and run-off
- valley infill swamps—occur in steeper topographies filling the valleys of incised second- or third-order streams. These swamps are more likely to be connected to either perched or regional aquifers
- hanging swamps—occur on steep valley sides where groundwater seepage is occurring.

Valley infill and hanging swamps are more vulnerable to subsidence impacts, since nonconventional subsidence affects cliffs and areas of steeper terrain rather than the flatter terrain where headwater swamps occur.

10.2 Ecological sensitivity modelling

For each conceptual model type, Bayesian belief networks (BBNs) were developed to model the sensitivity of the ecology of:

- ecological community as a whole, which was modelled assuming generic impact upon the interactions of each of the species
- giant burrowing frog, which was modelled recognising the specific habitat requirements of the species in relation to changes in fire and flow regimes, even where peat impacts remain stable
- Blue Mountains water skink, which was modelled recognising the specific habitat requirements of the species in relation to decreases in inundation and water quality impacts
- giant dragonfly, which was modelled recognising the specific habitat requirements of the species in relation to changes to peat stability and watertable stability, the latter of which affects the lifecycle of the species
- spreading rope rush, which was modelled recognising the specific habitat requirements of the species in relation to nutrient enrichment as a result of decreased inundation and decreased water quality, thereby increasing competition by invasive species.

The ecological community as a whole, and the individual species modelled were most sensitive to peat stability impacts, inundation and fire. Inundation also has a strong influence on peat stability and fire, so is overall the strongest influence on sensitivity for most species BBN models. Maintaining inundation is therefore critical to preventing impacts to the peat swamp. The type of conceptual model made no difference to the relative level of influence of the variables, with headwater, valley infill and hanging swamps all being the most sensitive to the same factors.

The giant dragonfly appears the worst affected in scenarios where subsidence impacts are severe and less severe. Therefore, at all levels of subsidence impact, the giant dragonfly is likely to be impacted. The Blue Mountains water skink, the giant burrowing frog and the peat swamp community as a whole are also greatly affected when subsidence impacts are severe, but demonstrate some tolerance to lower levels of impact. The spreading rope rush is more tolerant to subsidence impacts than the other species.

The BBN modelling is based on conceptualisation by specialists rather than on any measurement of impacts, since there is little information that specifically describes how ecology responds to changes in the surrounding environment. This means the BBNs should be used as a risk assessment tool, rather than a definitive measurement of impact. It is important to recognise this limitation of the model, and to use the BBN results to:

- flag the risk of potential impact to community/species
- indicate areas for priority investigation.

The primary use of the model is to design appropriate investigations to confirm the sensitivities suggested by the BBN, and to inform monitoring approaches in areas likely to be undermined. The BBN provides a framework that can be updated in the future, as empirical evidence of impacts to peat swamps becomes available.

10.3 Monitoring impacts on peat swamp ecology

A monitoring programme that aims to identify impacts early so that management can be adapted must focus on the subsidence or hydrological impacts, because these precede an ecological response. The time lag associated with ecological impacts means that monitoring ecology is not an acceptable method for early identification of impacts.

The multiple before–after control–impact (M-BACI) approach to monitoring swamp ecology is adopted here. It involves collecting data before and after impact occurs, from sites within the expected area of impact (impact swamps) and outside it (control swamps). Assuming all other natural variables are equal between the swamps at each measurement date, the change observed between control and impact sites after impact can be attributed to subsidence.

The recommended monitoring programme includes three separate phases:

- Phase 1—baseline characterisation of swamp ecology—to characterise the ecology of the peat swamps and to establish the natural range of variation in ecological variables before impact.
- Phase 2—risk assessment and agreement on acceptable levels of impact—to determine the risks for each swamp and define what an acceptable level of impact is, to inform the variables to be included in the ongoing monitoring programme.
- Phase 3—ongoing impact monitoring—to detect change in the variables that are expected to respond to subsidence impacts that cannot be attributed to natural variations.

Phases 1 and 3 incorporate field programmes, which should be designed according to the following steps: setting objectives, study design, field sampling, data analysis, data interpretation, reporting, and review and adaptation.

10.3.1 Baseline characterisation of swamp ecology

One of the key recommendations is the inclusion of an extensive baseline monitoring programme that involves collecting ecological field data for at least two years before mining, and analysis of remote-sensing data to extend the historical data period for as long as possible into the past. The baseline monitoring programme should include a broad suite of variables at a large number of swamps. The selection of the variables to include in the baseline monitoring programme should be informed by the expected ecological impacts (or ‘impact hypotheses’) developed at the beginning of the monitoring programme.

The methods recommended for baseline and impact monitoring should result in:

- documentation of flora and fauna species in the swamps, including identification of all threatened or vulnerable species and identification of invasive species
- detailed vegetation mapping of each swamp
- identification of the presence and abundance of threatened or vulnerable plants and animals in each swamp
- characterisation of swamp condition, including temporal variations in swamp extent and vegetation condition
- measurement of covariates/drivers of ecological response (i.e. hydrological regime).

Recommended methods are summarised in Table 10.1.

Table 10.1 Summary of methods recommended for baseline characterisation of swamp ecology.

Survey purpose	Survey method	Survey timing
Identification of flora species	Survey plots and species counts Random meanders	Twice a year in summer and winter
Vegetation mapping	Survey plots and species counts	At least 2 years before mining
Vegetation distribution	Survey plots and species counts Remote-sensing analysis using classification technique	Annually in spring or summer
Swamp extent	NDVI analysis of WorldView-2 or GeoEye Field GPS readings of swamp edges	Twice a year: in late summer where water availability is minimum and late winter when vegetation response to water availability is peak
Vegetation condition	NDVI analysis or unsupervised classification of WorldView-2or GeoEye imagery Field verification and visual inspection of natural colour aerial photography	Seasonally (4 times a year) for 5 years, then summer/winter for at least 10 years
Fauna species	Frogs: visual encounter surveys, audio strip transects, static call surveys and automated call recording Birds: diurnal surveys, call broadcast surveys and nest counts Invertebrates: diurnal searches for adults and exuviae Reptiles: diurnal hand searches and diurnal visual searches	Annual for all faunal groups, except birds which should be surveyed twice a year

10.3.2 Risk assessment and acceptable levels of impact

The risk assessment indicates the level of effort required and warranted to preserve each swamp. Defining the acceptable level of impact and the metrics by which this can be measured is critical for defining the ongoing monitoring programme, because it determines which variables need to be monitored, the frequency of monitoring (based on the ecological response of each variable) and optimum site selection for impact and control sites.

10.3.3 Ongoing impact monitoring programme

The suite of monitoring variables selected for ongoing monitoring should be based on the outcomes of phases 1 and 2, and should be those:

- that respond to specific disturbances related to longwall mining
- for which the range of natural variation is well established
- that monitor against the criteria agreed to measure acceptable levels of impact.

Detailed design of the monitoring programme relies on all earlier inputs but, in particular, the results of the baseline monitoring. Baseline monitoring data should be used to establish the spatial variability between swamps (both control and impact) and the temporal variability (over the 2-year field programme and the up to 20-year remote-sensing record). Understanding the variability enables control and impact sites to be selected that result in a statistically powerful analysis of impacts due to subsidence.

Implementation of the programme must be subject to review and adaptation to ensure that monitoring continues to focus on the most useful and responsive variables, and continued agreement of the level of acceptable impact to swamps.

The length of the ongoing monitoring period depends on the species present and their impact response times. If hydrological impacts have been observed, the ecological response may continue to occur for many years. Monitoring should continue until changes in ecology are no longer observed.

10.4 Knowledge gaps

The key knowledge gaps encountered in this study are lack of understanding of how swamp ecology responds to disturbances, and a lack of data on natural spatial and temporal variability in swamp ecology. Further information on these topics is needed to provide more specific advice on designing a programme for swamp monitoring, such as recommendations on monitoring variables and sampling frequency. Further research to address this knowledge gap would include a study to assess spatial variability between a large selection of swamps over a broad area, and temporal variability using field surveys and remote sensing. Other recommendations for further work include a pilot study to verify the most appropriate remote-sensing method (possible investigation the use of SEBAL to derive quantitative data on evapotranspiration) and integration of advice for developing a monitoring programme that includes ecological, hydrological and subsidence monitoring.

10.5 Future directions

To address the knowledge gaps in swamp variability and swamp ecological response to disturbance, a pilot research programme could be developed that compares ecological variation between impacted and non-impacted swamps. This would provide valuable information to help understand the existing spatial variability of the swamps.

The specific tasks should include:

- detailed vegetation mapping of each swamp selected for study (impacted and non-impacted)
- fauna surveys
- species abundance surveys.

The programme to establish knowledge on ecological variability between impacted and non-impacted sites must also consider the first- and second-order impacts of subsidence (rock deformation and hydrological impacts). Measurements of all three levels of impact need to coincide so that changes in rock integrity and hydrology can be linked to ecological response.

Remote sensing (with field verification) should be included in the programme, to extend the interpretations of swamp variability into the past. Remote-sensing data could be used to assess three variables over time for the selected swamps:

- changes in swamp extent
- changes in vegetation distribution within the swamp
- changes in vegetation health.

This programme would provide data to begin addressing the fundamental knowledge gap in the protection of peat swamps, which is how swamp ecology responds to disturbance. It would allow design of more effective monitoring programmes, and would provide information that could be used to develop adaptive management or remediation strategies for the swamps.

The data gained from such a programme should be used to refine the probabilities in the BBN. Adding quantified data to the BBN would strengthen its use as a risk assessment tool, making it more useful for predicting impacts on ecology for individual swamps in the early stages of the regulatory approvals process. Depending on the confidence in the quantified data, it may also help define trigger levels for ecological impact in terms of inundation thresholds required to maintain particular species and, potentially, the ecological community overall.

Peer review comments: additional research needed

A number of comments from peer reviewers and NSW Government agencies focused on the need for the recommendations for ecological monitoring provided in this report to be accompanied by a robust hydrological monitoring programme (see comments at the end of Chapter 8). In addition, monitoring programmes designed around geological (subsidence or rock fracturing) and geomorphic processes have been suggested as integral to any comprehensive monitoring regime. Additional references have been added in the References, under ‘Additional references provided by peer reviewers’.

Appendix A: Bayesian belief networks and definitions

This section describes the methodology behind the construction of Bayesian belief networks (BBNs), how the information was obtained and used to populate the models, and how the sensitivity analysis was achieved.

As described in Chapter 5, BBNs were used to model the impacts of longwall coalmining on Temperate Highland Peat Swamps on Sandstone (THPSS) communities.

Expert elicitation was used to model the complex relationships that describe the function of the peat swamps.

A1 Model development

A1.1 Development of baseline model structure

After the development of the conceptual models for peat swamps (described in Chapter 3), a brainstorming session produced a draft BBN that was used as a baseline model for the planned expert workshop. The model incorporated the areas of ecology, hydrology, geology, hydrogeology and mining design characteristics.

A specialist workshop was run in Sydney on 24 October 2012. A range of specialists participated in the workshops, including hydrologists, ecologists, hydrogeologists and mining representatives. Specialists were asked to critique the baseline model structure and change it based on their knowledge of the relationships that describe peat swamp function. The specialists were then asked to identify states for each node, discuss definitions of each node and state, and to populate the probabilities under each child node. The BBN was constructed so the first state listed under the node was the most likely to result in impact and the last state was the least likely to impact the peat swamp.

The baseline model structure developed in consultation with the specialists is shown in Figure 5.2. The arrows between nodes indicate how changes in one node influence the next. Based on the literature and specialist knowledge, each node in the BBN is considered to be an important factor in the function and maintenance of the peat swamps. Definitions for each node and state are given in Section A4.

Waste water discharge from mines to the swamps has not been considered in the BBN because it was considered that this (if it occurred) would occur further downstream in the catchment than the peat swamps. Waste water discharge from mines has occurred above peat swamps in the past, but alternative discharge options are now in place, and it is unlikely that discharge upstream of swamps would be allowed again (M Krogh, 2012, pers. comm., 3 December). Compared with the risks to the swamps from subsidence, waste water releases are a low risk and have been excluded from the BBN for simplicity. Other threats that are not directly related to longwall mining have also not been considered in the BBN, such as weed invasion, evaporation, land clearing, agriculture, groundwater extraction, or disturbances from above groundwater infrastructure such as roads, buildings, reservoirs or mine pits.

The software Netica 4.12 (Norsys Software Corp 2009) was used to develop and compile the Bayesian nets once the probabilities under each child node were populated and sensitivity analysis was run for each child node.

A1.2 Development of BBNs for conceptual models, the swamp community and species

Conceptualisation of the peat swamp and surrounding landscape identified three conceptual models:

- headwater swamps
- valley infill swamps
- hanging swamps.

To distinguish the steeper topography and groundwater connection of hanging swamps and valley infill swamps from the flat topography and lack of groundwater connection for headwater swamps, a different set of probabilities was assigned for selected nodes. This effectively resulted in the development of two BBNs that described the three conceptual models:

- BBN1—for hanging swamps and valley infill swamps, with probabilities assigned to recognise the greater susceptibility to subsidence impacts engendered by the steeper topography and groundwater connection
- BBN2—for headwater swamps, where probabilities reflected the lower vulnerability of headwater swamps due to flat, elevated topography and lack of groundwater connection.

The key understanding required to develop the model structure is how the various physical landscape components interact, and how longwall coalmining is likely to alter these interactions. As an example, it is known that groundwater flow to swamps occurs mainly through fractures, joints and bedding planes in the sandstone aquifers (Young 2007). Longwall mining can cause fracturing of the sandstone at the surface near the base of the peat swamp. The resulting increase in permeability alters the groundwater flow to the swamp, and is most likely to reduce groundwater discharge to the swamp. Similarly, it is known that swamps need a relatively stable level of inundation to avoid peat drying, shrinkage and erosion. Where subsidence occurs below a peat swamp, the increase in permeability can cause the swamp to drain, thereby causing the swamp to dry out and destabilising the peaty substrate. These are the types of relationships that are captured in the BBN.

To try to better understand the functioning of the peat swamp ecosystem modelling was done for the sensitivity of the peat swamp community as a whole, as well as for individual species within the peat swamps., There were some species for which habitat requirements had been described in sufficient detail to use in the BBN. These are:

- THPSS ecological community
- giant burrowing frog
- Blue Mountains water skink
- giant dragonfly
- spreading rope rush.

An individual BBN was developed to model each species and the ecosystem as a whole. The probability of impact for these species varied depending on their specific habitat requirements. Table A1 summarises the models developed to assess the sensitivity of these species and the ecosystem for each conceptual model type.

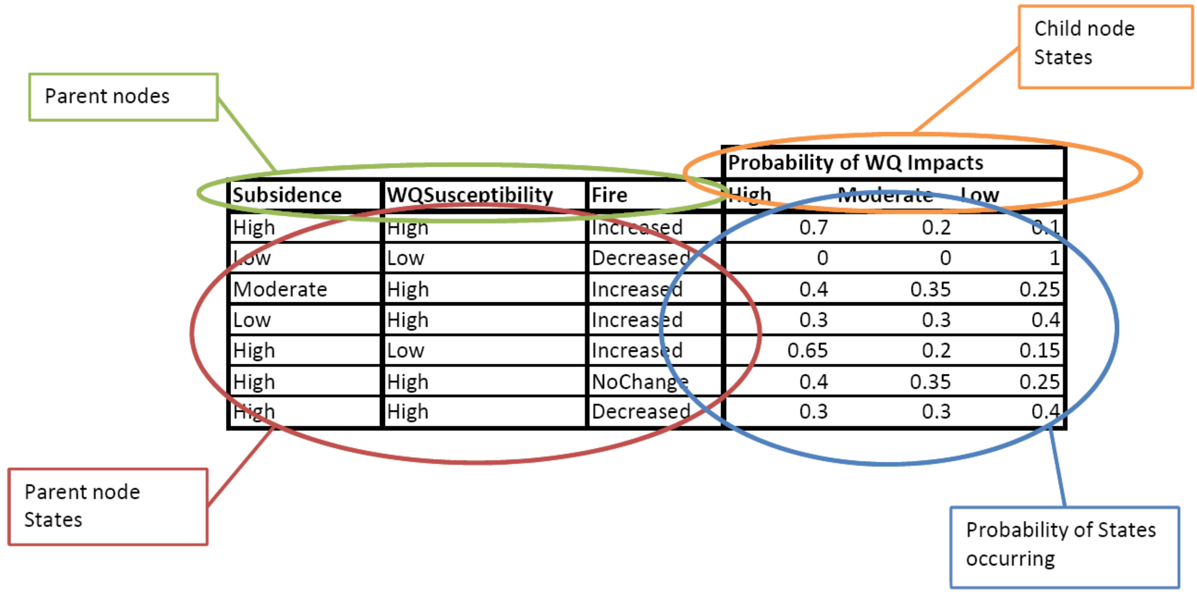
Table A1 Summary of BBNs developed to capture each conceptual model and each community/species.

Conceptual model	Community/Species				
	Threatened ecological community	Giant burrowing frog	Blue Mountains water skink	Giant dragonfly	Spreading rope rush
Hanging and valley infill swamps	✓	✓	✓	✓	✓
Headwater swamps	✓	✓	✓	✓	✓

A1.3 Approach to assigning probabilities for individual models

Bayesian networks are reliant on the conditional probability tables under each child node. Probabilities were assigned by the experts at the workshop and confirmed by further discussions with hydrogeologists and ecologists using phone interviews.

The probability tables are made up of every combination of state for each node pointing towards the child node. To assign probabilities, the combination of states in the probability table is reviewed, and given that combination of states, the likelihood of each state in the child node occurring is assigned a probability (Figure A1). The probabilities that link the nodes were informed by the literature and specialist knowledge. Quantified data (i.e. from numerical modelling or field trials) was generally not available to inform the probabilities. However, useful qualitative descriptions were given in the literature that described the impacts on the physical environment caused by longwall mining. This qualitative data and specialists' knowledge was sufficient to assign numbers (probabilities) to the relationships in the BBN.



Subsidence	WQSusceptibility	Fire	Probability of WQ Impacts		
			High	Moderate	Low
High	High	Increased	0.7	0.2	0.1
Low	Low	Decreased	0	0	1
Moderate	High	Increased	0.4	0.35	0.25
Low	High	Increased	0.3	0.3	0.4
High	Low	Increased	0.65	0.2	0.15
High	High	NoChange	0.4	0.35	0.25
High	High	Decreased	0.3	0.3	0.4

Figure A1 Example of a conditional probability table, where probabilities are entered to describe the likelihood of the state in the child node occurring given the combination of states in the parent nodes.

Probabilities were varied between the conceptual models as follows:

- The probability that ‘change in sandstone permeability’ influences ‘change in groundwater connection’ was relatively high for hanging swamps and valley infill swamps, since groundwater connection contributes to swamp hydrology for these conceptual models. For headwater swamps, there is a zero probability that ‘change in sandstone permeability’ influences ‘change in groundwater connection’, since headwater swamps are not connected to groundwater.
- The probabilities for each state in the parent node ‘channel incision’ were varied for the conceptual models because valley infill and hanging swamps occur in more incised topographies (so that states ‘incised’ and ‘moderate’ were weighted more heavily), and headwater swamps occur in flat areas, so the state ‘flat’ was weighted the heaviest.

Some of the combinations of states in the probability tables are not possible as they cannot possibly occur within the environment. In this case, a state ‘impossible’ was added to the child node and the probabilities were given according to these circumstances. An example of this is the relationship between ‘fire’ and ‘inundation’. The change in frequency and intensity of fires burning through the peat swamps is solely controlled by the inundation of the swamp: where inundation decreases, the swamp is drier and is therefore more likely to experience fires of increased intensity and frequency. Similarly, a wetter swamp (where inundation has increased) is more protected from fire. The inverse situation, where inundation increases and fire frequency and intensity also increase, cannot occur. Situations like this were assigned as ‘impossible’ in the BBN probability tables.

A1.3.1 Ecological impact nodes

Information linking subsidence effects to ecological impacts is limited, with little information that specifically describes how ecology responds to changes in the surrounding environment. The lack of data necessitated a pragmatic and simplified approach to assigning probabilities to the ecological impact nodes. This approach was not used to assign probabilities to other nodes in the model because the interaction between other nodes is better understood (i.e. has been modelled, observed, reported).

The simplified approach for identifying impacts to the community/species means the BBNs should be used as a risk assessment tool, rather than a definitive measurement of impact. It is important to recognise this limitation of the model and to use the BBN results to:

- flag the risk of potential impact to community/species
- indicate areas for priority investigation.

The primary use of the model should therefore be to design appropriate investigations to confirm the sensitivities suggested by the BBN and to inform monitoring approaches in areas likely to be undermined.

The factors controlling ecological sensitivity are different for each community/species, depending on the habitat requirements of each community/species. As such, the probabilities that reflect the likely impact of changes in the physical environment on the ecology were varied for each community/species. Modelling the probability of impact to the ecological community as a whole used a generic understanding of ecosystem response, whereas modelling individual species used knowledge of specific habitat requirements and how subsidence was likely to alter the habitat.

More sensitive species were assigned probabilities that indicate a higher likelihood of high impact, whereas the probabilities for more resilient species reflected a lower likelihood of impact. The logic for assigning probabilities for each species are described below.

Giant burrowing frog (*Heleioporus australiacus*)

Biophysical elements of the peat swamp environment that are likely to highly impact the viability of the giant burrowing frog include changes to fire and flow, even where peat remains stable. This is likely to be a key interaction for the species. Although the species has fairly diverse habitat requirements, from slow-flowing streams to dams and swamps, a decreased fire risk but high change in flow still represent a high risk to the species, especially an increase in flow that might reduce low flow and pool habitats. Minor changes to flow regimes and a decreased risk of fire were considered likely to have a low impact on the viability of the species, even if peat instability occurs. Elements likely to have little influence on the viability of the species include minor changes to flow. Low sensitivity to some vegetation elements in the swamp environment may mean some habitat is retained, even with increased fire and flow regime change, representing some risk reduction.

Blue Mountains water skink (*Eulamprus leuraensis*)

The Blue Mountains water skink inhabits boggy soils that are permanently wet. Larger, wetter swamps close together are more likely to support skinks than small, dry swamps isolated from each other. Critical requirements for maintaining the viability of skink populations in the swamp environment include stable watertables and water quality. High probabilities of impacts to the species occur when inundation is decreased and decreases in water quality impact on food resources. A key interaction affecting the viability of the species is changes to vegetation as a result of subsidence resulting in loss of vegetation for the species. The logic for assigning probabilities for impact to the Blue Mountains water skink in the BBN was therefore:

- Any decrease in inundation was assumed to have a high impact on the water skink.

Giant dragonfly (*Petalura gigantea*)

Critical habitat requirements for maintaining the viability of giant dragonfly populations in the swamp environment include stable peat environments and stable watertables. Key threats to these requirements include any activity that causes peat instability resulting in destruction of nymph burrows and lifecycle interruption. Changes in watertable levels that could expose or drown burrows, and loss of surface water (especially in summer)—which may limit foraging opportunities by adults—are also considered key threats to critical habitat requirements of the species. Biophysical elements of the peat swamp environment that are therefore likely to highly impact the viability of the giant dragonfly include high peat instability regardless of other factors, changes in inundation regardless of other factors and high vegetation sensitivity. The logic for assigning probabilities for impact to the giant dragonfly in the BBN was therefore:

- where peat stability was high, impact on the giant dragonfly was always assumed to be high
- any decrease in inundation was assumed to have a high impact on the giant dragonfly
- vegetation susceptibility influences impact where peat stability and/or changes to inundation are not high.

Spreading rope rush (*Empodisma minus*)

Based on the literature about its ecology, spreading rope rush does not appear to be highly sensitive. Biophysical elements of the peat swamp environment that are likely to highly impact the viability of the species include water quality and inundation changes that may impact on nutrient enrichment of the swamp. This is likely to be a key interaction for the species. Minor changes to flow regimes and a decreased risk of fire were considered likely to have a moderate impact to the viability of the species. Elements likely to have little influence on the viability of the species include peat stability impacts and fire. The species is a peat creator and so is likely to be an important element in the maintenance of swampland ecological integrity in the face of mitigating longwall mining impacts. The 'vegetation type' node was removed from the spreading rope rush model, since the viability of the spreading rope rush does not depend on the presence of other vegetation.

A1.4 Model behaviour and sensitivity analysis

Feedback on the model was sought on two occasions: where experts involved in the workshop were encouraged to critically analyse the completed models via email and later through phone interviews to edit any of the conditional probability tables if they were not performing as intended. The model was validated using sensitivity analysis. Sensitivity to findings was determined by running sensitivity analysis within Netica to determine the effect of each parameter on ecological community change. Sensitivity analysis in Netica determines how 'sensitive' a model is to changes in model parameters. By measuring the uncertainty in the model, emphasis can be placed on parameters with enough sensitivity to significantly affect the model behaviour when parameter values are changed. We determined the entropy reduction (variance reduction), which is the expected reduction in uncertainty of the node being queried (e.g. peat swamp stability) due to information being given at the parent node (e.g. inundation). Hence, if information is supplied about the state of a parent node, this may reduce the maximum range of values possible in the distribution of the output node and reduce its uncertainty and variance within the distribution (Norsys Software Corp 2009; Nash et al. 2010).

The sensitivity of the nodes captures the influence of the different variables on the peat swamp community and the individual species.

A2 Results

A2.1 Model outputs

A BBN model with 10 input variables (parent nodes) showing a directional relationship to 11 linked nodes (child nodes) was developed. The definitions for each node and state are listed in Section A5. The BBN modelled a change in the ecological community as the overall resultant impact on peat swamps. A change in the ecological community was affected by peat stability, water quality, inundation, fire and flow regime. Peat stability was in turn influenced by flow regime, inundation, fire and subsidence. Subsidence impacts were related to the geological characteristics of the area, mine dimensions, channel incision, mining depth and proximity of the mining activity. In turn, subsidence affected inundation, permeability of the rock, the flow regime, water quality and peat stability.

The peat swamps were represented in two models: hanging swamps and valley infill swamps (Figure A2) and headwater swamps (Figure A3). These represent the three conceptual models for peat swamps that behave slightly differently. The main differences between BBNs were the channel incision and change in groundwater connection nodes. Hanging swamps and valley infill swamps occur in more incised topographies and are therefore more susceptible to subsidence impacts. Headwater swamps tend to occur in flatter topographies and the general absence of channel incision results in lower impacts from subsidence. Headwater swamps are also unaffected by changes in groundwater connection, since they are generally not connected to groundwater. Conversely, hanging and valley infill swamps are reliant on groundwater discharge and are therefore affected by changes in groundwater connection.

The two models were developed to ascertain any changes in the overall ecological community of the peat swamp as a generic community model. The models were then run to determine the individual impacts on one flora species (spreading rope rush, *Empodisma minus*) and three fauna species. The fauna species modelled were the giant dragonfly (*Petalura gigantea*), the giant burrowing frog (*Helioporos australiacus*) and the Blue Mountains water skink (*Eulamprus leuraensis*).

There was a strong dependency between inundation and fire, with fire only being affected if inundation increased (fire risk decreased), inundation decreased (fire risk increased) or when there was no change in both. All other combinations were deemed impossible, such as increased inundation and increased fire risk. These events were captured by the impossible state in the peat stability impact and the ecological community change nodes.

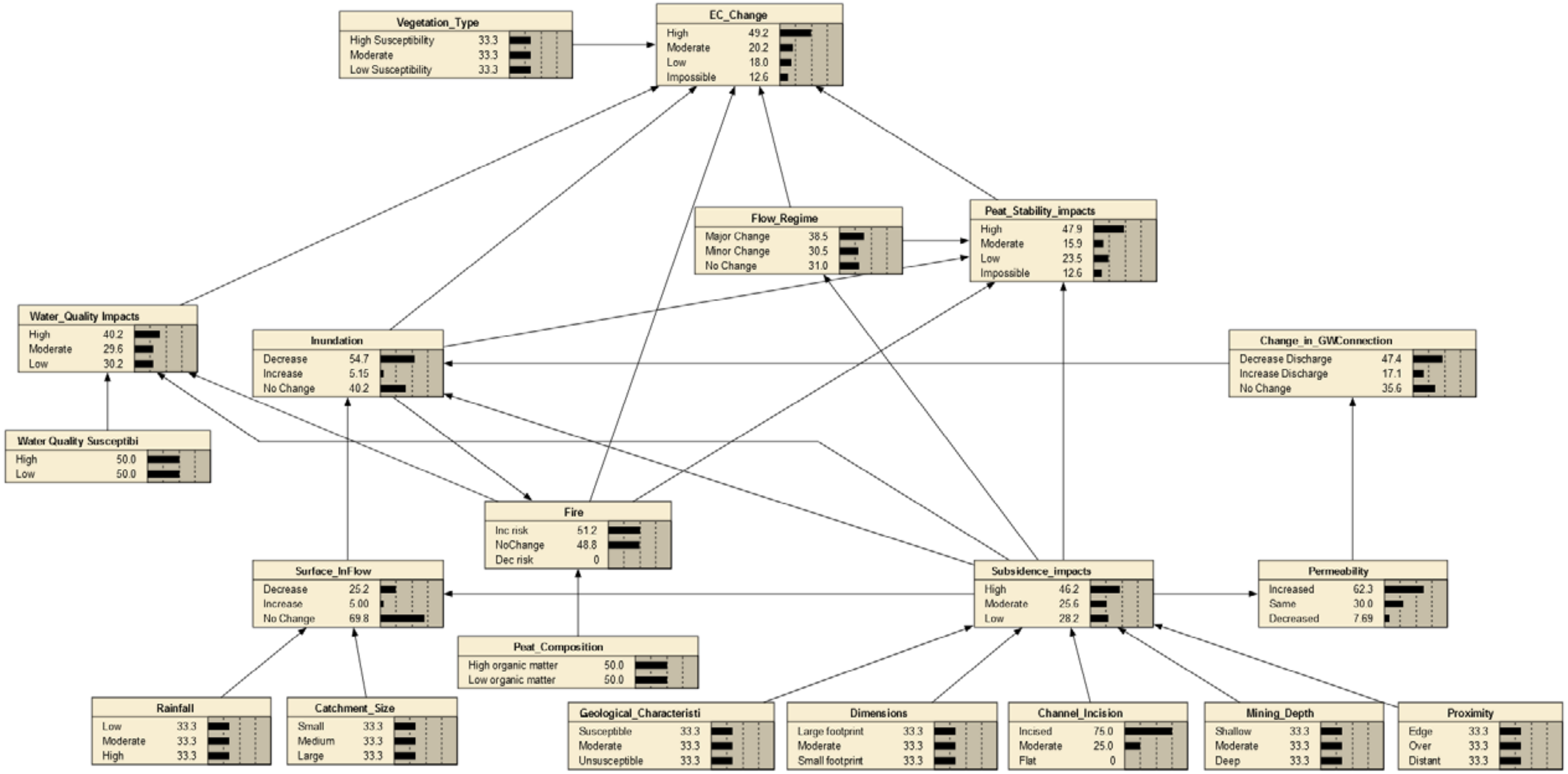


Figure A2 Bayesian belief network of general model of change in ecosystem condition for hanging swamps and valley infill swamps.

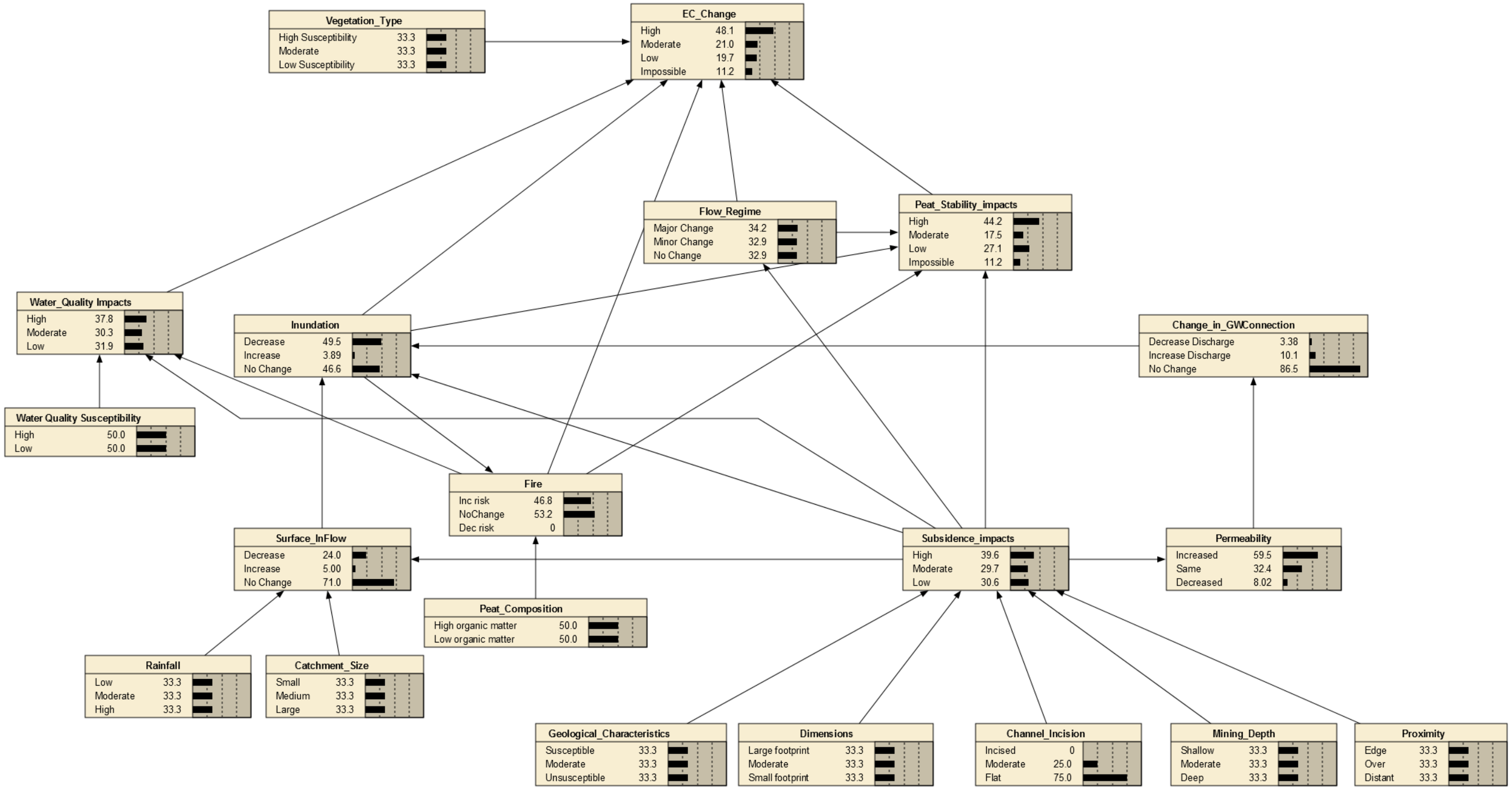


Figure A3 Bayesian belief network of general model of change in ecosystem condition for headwater swamps.

A2.2 Sensitivity analysis

A2.2.1 Ecological community model

Results from modelling the overall ecological community to the effects of subsidence are shown in Figure A4, which shows the influence of the parent nodes on each child node. The actual values of entropy reduction (on the x axis in Figure A4) are not comparable between the different child nodes; however, for a single child node the entropy reduction values show the relative influence of each parent node. This indicates which physical changes will have the greatest effect on the peat swamps, as defined by the probabilities assigned during model development. It therefore identifies the physical change to the environment to which the peat swamp is expected to be most sensitive.

The graph only displays the direct parents of each child node. A parent node may also have an indirect influence on a node through another node. For example, in the baseline model inundation affects ‘change in ecological community’ both directly and indirectly through ‘impacts on peat stability’. Indirect nodes can still have influence through the intermediate child node but the influence reduces with distance from the target node.

The results of the sensitivity analysis are discussed in Section 5.3.1; however, in summary, both the headwater swamp model and the hanging and valley infill swamps model, change to the ecological community was strongly influenced by the stability of the peat, followed by a lesser effect from inundation and fire. This highlights that erosion of the peat has catastrophic impacts on the health of the peat swamp. The impact of fire is fully dependent on the level of inundation, since the wetness of the swamp controls the frequency and intensity of fires that burn through the swamp. Therefore, the influence of these nodes will always be similar.

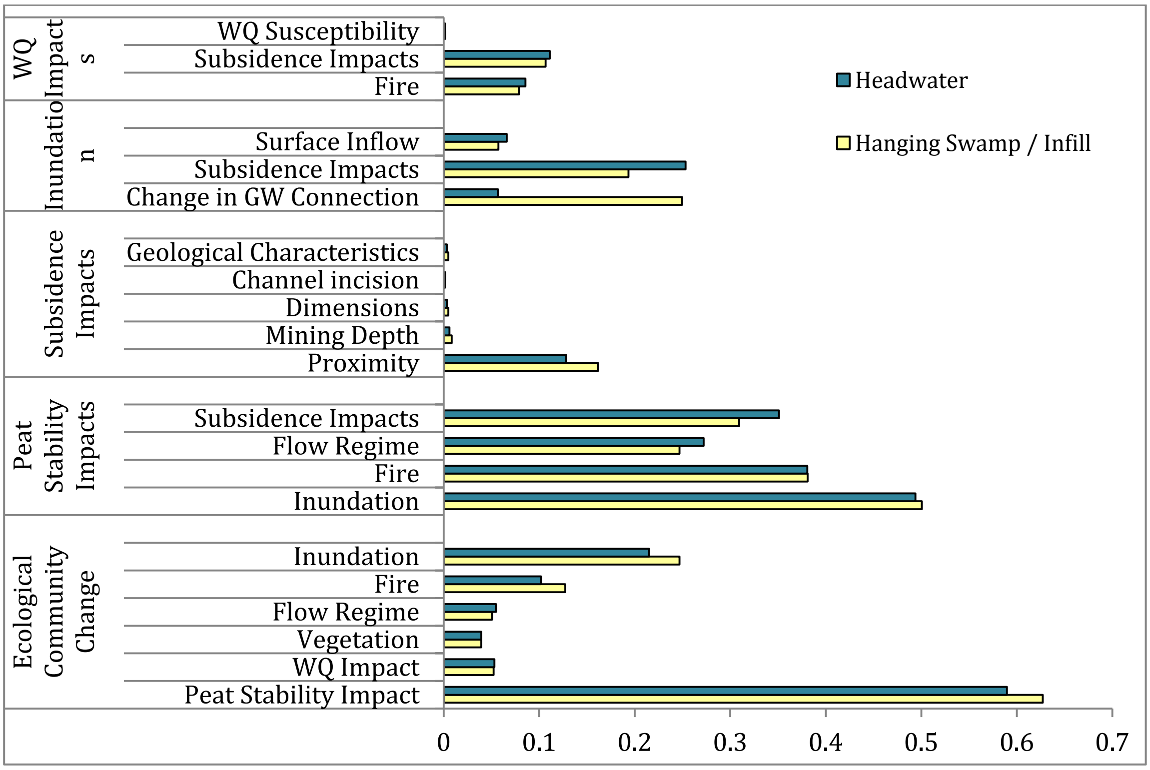


Figure A4 Results of the sensitivity analysis showing the entropy or variance reduction for the community models for both hanging and valley infill swamps and headwater swamps. The greater the value, the more influence the variable had on the model.

A2.2.2 Individual species models

For the modelling of individual species, the only node that varied between the BBNs was 'change in the ecological community'. For each individual species model, the ecological community change node was changed to the particular species being modelled. Results for the modelling of the ecological community as a whole and the individual species are discussed in Section 5.3.2 and shown in Figure 5.3. The results for the THPSS ecological community and the individual species to each physical factor modelled in the BBNs are also summarised in Figure A5 and Table A2.

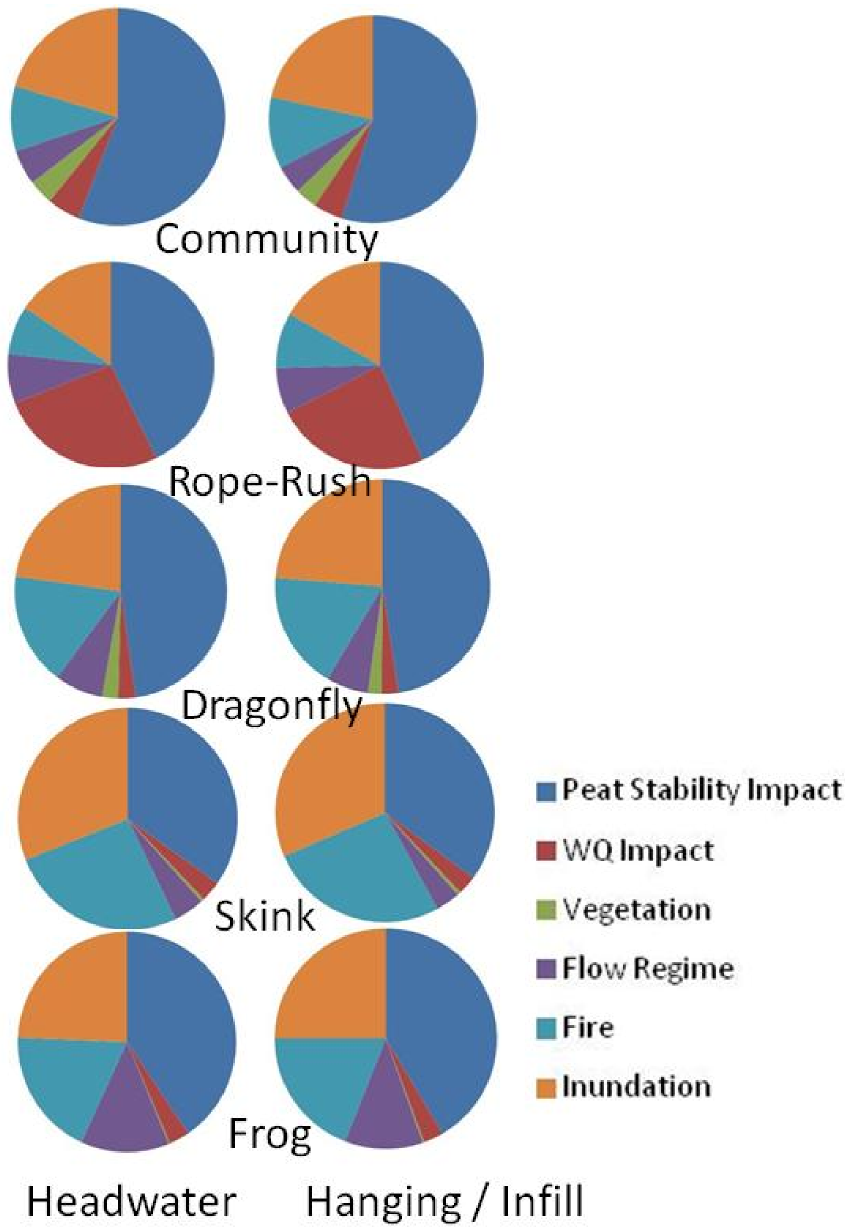


Figure A5 Relative influence of each parent node on the swamp types and community/species modelled in the BBNs.

Table A2 Summary of the relative sensitivity of the community and each species to each physical factor.

Conceptual model	Community/Species				
	Threatened ecological community	**Giant burrowing frog**	**Blue Mountains water skink**	**Giant dragonfly**	**Spreading rope rush**
Hanging/valley infill swamps	Peat stability Inundation Fire Water quality Flow regime Vegetation	Peat stability Inundation Fire Flow regime Water quality Vegetation	Peat stability Inundation Fire Flow regime Water quality Vegetation	Peat stability Inundation Fire Flow regime Water quality Vegetation	Peat stability Water quality Inundation Flow regime
Headwater swamps	Peat stability Inundation Fire Water quality Flow regime Vegetation	Peat stability Inundation Fire Flow regime Water quality Vegetation	Peat stability Inundation Fire Flow regime Water quality Vegetation	Peat stability Inundation Fire Flow regime Water quality Vegetation	Peat stability Water quality Inundation Flow regime Fire

The type of conceptual model made no difference to the relative level of influence of the variables, with headwater and hanging/valley infill swamps being the most sensitive to the same parent nodes.

In summary, peat swamps generally have the highest sensitivity to changes in peat stability. However, peat stability is strongly influenced by inundation, because a decrease in inundation can cause drying, cracking and erosion of the peat. Maintaining inundation is therefore critical to preventing impacts on the peat swamp.

A2.3 Scenario analysis

The parent nodes for each model are water quality susceptibility, rainfall, catchment size, peat composition, geological characteristics, dimensions, channel incision, mining depth and proximity. The probabilities in these parent nodes can be changed to reflect actual conditions. For example, the probabilities can be changed so that the states reflect the actual rainfall, catchment size, peat composition or channel incision of a particular swamp. Similarly, the probabilities in the parent nodes can be changed to reflect different impact scenarios. That is, if in each parent node, the states that are the most detrimental to the swamps were given the highest probability, the model results would show the worst-case impacts for the swamps (high-impact scenario). Conversely, if the states in each parent node that were the least detrimental were given the highest probability, the model results would show the best-case impact for the swamps (low-impact scenario). The states in each parent node can be weighted differently (given different probabilities) to reflect worst- or best-case scenarios to indicate a range of predicted impacts on the peat swamps, or they can be weighted to reflect actual conditions for a particular peat swamp. High-impact and low-impact scenarios are discussed below.

The probability of high impact for inundation, peat stability and subsidence had identical values across the community and fauna models. This pattern was repeated when the lowest impact scenario was run, although the values were reduced. Hanging/infill swamps had slightly higher probabilities of impact over headwater swamps throughout the model, since changes in groundwater and steeper topography increase impacts. The probabilities describing the effect of the parent nodes varied for the ‘change in the ecological community’ node across the different species and swamps, and between the different scenarios. This reflects the varying sensitivity of the community and the individual species to changes in the physical environment. Table A3 summarises the impacts for low and high-impact scenarios.

A2.3.1 High-impact scenario

If the factors input into the model (e.g. mining depth and catchment size) were the most detrimental for the community or species, it is expected the probability of impact would be high. The giant dragonfly would be the most impacted (87.5 per cent for infill swamps and 85.8 per cent for headwater swamps) of all the individual species in the high-impact scenario, reflecting its greater sensitivity to change. The giant burrowing frog also was highly impacted (84.1 per cent for infill swamps and 80.9 per cent for headwater swamps). The impacts were also high for the community model (79.9 per cent for infill swamps and 78.4 per cent for headwater swamps). The impact on the water skink was less (77.8 per cent for infill and 74.7 per cent for headwater) but it would be still detrimental to this threatened species if the scenario states occurred.

Generally, the level of impact for the community as a whole and for the fauna species was high, indicating that significant changes to the swamp environment have a high probability of high impact on swamp ecology. The rope rush was much more robust against high impact (42.0 per cent for the infill swamps and 40.4 per cent for the headwater swamps) compared with the fauna species. While the spreading rope rush was less affected, over time the result may be significant.

These values show a combination of highly impacting factors will be detrimental to all the faunal species modelled. The proximity of the mine was one of the major indirect influences for high impact. If proximity was changed from edge to distant, the overall impact on the dragonfly changed from 87.5 per cent high impact to 62.5 per cent high impact. The other factors that changed these values more than a few per cent when changed to low-impact states were vegetation type (down to 54.6 per cent) and peat composition (down to 47.8 per cent. Proximity affects subsidence impacts significantly, which in turn affects a number of direct and indirect nodes, such as inundation and peat stability. These are the key drivers determining impact on each species and the community. Peat composition affects fire, which in turn affects peat stability and the ecological community of the swamp. The susceptibility of vegetation type only affects the latter.

A2.3.2 Low-impact scenario

If the factors input into the model (e.g. mining depth and catchment size) were the least detrimental for the community or species, it is expected the probability of impact would be low. The water skink would be the least impacted fauna species in this case (25.6 per cent for infill swamps and 17.8 per cent for headwater swamps), reflecting its greater resilience compared with the other fauna species modelled. The spreading rope rush was the least impacted overall (19.2 per cent for infill swamps and 17.6 per cent for headwater swamps) in the low-impact scenario. There was a similar lower impact on the community model for the infill swamps (26 per cent) and for the headwater swamps (22.7 per cent) in the low-impact scenario. The effect of low-impact factors on the burrowing frog was increased slightly in the headwater (24.1per cent) and infill swamps (32.7 per cent). The giant dragonfly was the most affected in the low-impact scenario with 43.9 per cent for the infill swamps and 35.1 per cent for the headwater swamps.

Few data are available on how changes to hydrology specifically impacts flora and fauna. Anecdotal evidence suggests species composition change, species health deterioration and fauna extinction due to habitat destruction will occur.

Results of the low-impact scenario analysis showed that if precautions were taken to minimise impact on the peat swamp community, the water skink and the burrowing frog would benefit the most and the rope rush and giant dragonfly the least. This is because even under a low-impact scenario, the dragonfly is still very sensitive to any change in the peat swamp environment. For the rope rush, the opposite is true: it is comparatively resilient, and therefore there is less difference in impact between the high and low-impact scenarios than there is for the other species modelled. The rope rush shows the least impact overall and is expected to be more robust to change, at least in the short term. Proximity is again the biggest contributor to impact change, with the skink changing from 25.6 per cent to 37.5 per cent at the high state and the dragonfly changing from a high 43.9 per cent to 59.7 per cent at the high state.

The scenario analyses showed the peat swamp ecological community and the individual modelled species were all impacted by the mining effects and associated ecosystem changes. The giant dragonfly appears the worst affected at high impacts but is also substantially affected with low impact. The water skink and the burrowing are also greatly affected when impacts are high, but these levels can be reduced if the impacts are lowered. The peat swamp community had a similar effect. The rope rush is not as highly impacted as the other species.

Table A3 Results of the scenario analysis for the community model and the individual species model for the hanging and infill swamps and for the headwater swamps.

Model	Scenario	Node	States (%)							
Infill	High impact	EC_Change	High	79.9	Moderate	8.08	Low	5.07	Impossible	6.92
		Peat stability impacts	High	78.2	Moderate	9.67	Low	5.18	Impossible	6.92
		Inundation	Decrease	75.5	Increase	4.96	No change	19.6		
		Subsidence impacts	High	95.0	Moderate	5.0	Low	0		
	Low impact	EC_Change	High	26.0	Moderate	24.4	Low	36.7	Impossible	12.9
		Peat stability impacts	High	26.9	Moderate	18.2	Low	42.0	Impossible	12.9
		Inundation	Decrease	36.4	Increase	5.66	No change	58.0		
		Subsidence impacts	High	14.0	Moderate	30.0	Low	56.0		
Headwater	High impact	EC_Change	High	78.4	Moderate	9.53	Low	6.28	Impossible	5.75
		Peat stability impacts	High	74.2	Moderate	11.3	Low	8.77	Impossible	5.75
		Inundation	Decrease	72.2	Increase	3.30	No change	24.5		
		Subsidence impacts	High	85.0	Moderate	10.0	Low	5.0		
	Low impact	EC_Change	High	22.7	Moderate	23.4	Low	44.0	Impossible	9.91
		Peat stability impacts	High	19.8	Moderate	17.2	Low	53.1	Impossible	9.91
		Inundation	Decrease	23.4	Increase	5.23	No change	71.3		
		Subsidence impacts	High	10.0	Moderate	20.0	Low	70.0		
Spreading rope rush—infill	High impact (moderate vegetation)	EC_Change	High	42.0	Moderate	36.1	Low	14.9	Impossible	6.92
		Peat stability impacts	High	78.2	Moderate	9.67	Low	5.18	Impossible	6.92
		Inundation	Decrease	75.5	Increase	4.96	No change	19.6		
		Subsidence impacts	High	95.0	Moderate	5.0	Low	0		

Model	Scenario	Node	States (%)							
	Low impact	EC_Change	High	19.2	Moderate	21.0	Low	46.9	Impossible	12.9
		Peat stability impacts	High	26.9	Moderate	18.2	Low	42	Impossible	12.9
		Inundation	Decrease	36.4	Increase	5.66	No change	58		
		Subsidence impacts	High	14.0	Moderate	30	Low	56		
Spreading rope rush—headwater	High impact (moderate vegetation)	EC_Change	High	40.4	Moderate	35.4	Low	18.5	Impossible	5.75
		Peat stability impacts	High	74.2	Moderate	11.3	Low	8.77	Impossible	5.75
		Inundation	Decrease	72.2	Increase	3.30	No change	24.5		
		Subsidence impacts	High	85.0	Moderate	10.0	Low	5.0		
	Low impact	EC_Change	High	17.6	Moderate	19.3	Low	53.2	Impossible	9.91
		Peat stability impacts	High	19.8	Moderate	17.2	Low	53.1	Impossible	9.91
		Inundation	Decrease	23.4	Increase	5.23	No change	71.3		
		Subsidence impacts	High	10.0	Moderate	20.0	Low	70.0		
Giant dragonfly—infill	High impact	EC_Change	High	87.5	Moderate	4.47	Low	1.13	Impossible	6.92
		Peat stability impacts	High	78.2	Moderate	9.67	Low	5.18	Impossible	6.92
		Inundation	Decrease	75.5	Increase	4.96	No change	19.6		
		Subsidence impacts	High	95.0	Moderate	5.0	Low	0		
	Low impact	EC_Change	High	43.9	Moderate	8.26	Low	34.9	Impossible	12.9
		Peat stability impacts	High	26.9	Moderate	18.2	Low	42.0	Impossible	12.9
		Inundation	Decrease	36.4	Increase	5.66	No change	58.0		
		Subsidence impacts	High	14.0	Moderate	30.0	Low	56.0		
Giant dragonfly—	High impact	EC_Change	High	85.8	Moderate	6.87	Low	1.59	Impossible	5.75
		Peat stability impacts	High	74.2	Moderate	11.3	Low	8.77	Impossible	5.75

Model	Scenario	Node	States (%)							
headwater		Inundation	Decrease	72.2	Increase	3.30	No change	24.5		
		Subsidence impacts	High	85.0	Moderate	10.0	Low	5.0		
	Low impact	EC_Change	High	35.1	Moderate	9.31	Low	45.7	Impossible	9.91
		Peat stability impacts	High	19.8	Moderate	17.2	Low	53.1	Impossible	9.91
		Inundation	Decrease	23.4	Increase	5.23	No change	71.3		
		Subsidence impacts	High	10.0	Moderate	20.0	Low	70.0		
Skink—infill	High impact	EC_Change	High	77.8	Moderate	2.98	Low	12.3	Impossible	6.92
		Peat stability impacts	High	78.2	Moderate	9.67	Low	5.18	Impossible	6.92
		Inundation	Decrease	75.5	Increase	4.96	No change	19.6		
		Subsidence impacts	High	95.0	Moderate	5.0	Low	0		
	Low impact	EC_Change	High	25.6	Moderate	7.96	Low	53.5	Impossible	12.9
		Peat stability impacts	High	26.9	Moderate	18.2	Low	42.0	Impossible	12.9
		Inundation	Decrease	36.4	Increase	5.66	No change	58.0		
		Subsidence impacts	High	14.0	Moderate	30.0	Low	56.0		
Skink—headwater	High impact	EC_Change	High	74.7	Moderate	3.37	Low	16.1	Impossible	5.75
		Peat stability impacts	High	74.2	Moderate	11.3	Low	8.77	Impossible	5.75
		Inundation	Decrease	72.2	Increase	3.30	No change	24.5		
		Subsidence impacts	High	85.0	Moderate	10.0	Low	5.0		
	Low impact	EC_Change	High	17.8	Moderate	6.88	Low	65.5	Impossible	9.91
		Peat stability impacts	High	19.8	Moderate	17.2	Low	53.1	Impossible	9.91
		Inundation	Decrease	23.4	Increase	5.23	No change	71.3		
		Subsidence impacts	High	10.0	Moderate	20.0	Low	70.0		

Model	Scenario	Node	States (%)							
Frog—infill	High impact	EC_Change	High	84.1	Moderate	4.57	Low	4.40	Impossible	6.92
		Peat stability impacts	High	78.2	Moderate	9.67	Low	5.18	Impossible	6.92
		Inundation	Decrease	75.5	Increase	4.96	No change	19.6		
		Subsidence impacts	High	95.0	Moderate	5.0	Low	0		
	Low impact	EC_Change	High	32.7	Moderate	10.5	Low	43.9	Impossible	12.9
		Peat stability impacts	High	26.9	Moderate	18.2	Low	42.0	Impossible	12.9
		Inundation	Decrease	36.4	Increase	5.66	No change	58.0		
		Subsidence impacts	High	14.0	Moderate	30.0	Low	56.0		
Frog—headwater	High impact	EC_Change	High	80.9	Moderate	5.35	Low	8.03	Impossible	5.75
		Peat stability impacts	High	74.2	Moderate	11.3	Low	8.77	Impossible	5.75
		Inundation	Decrease	72.2	Increase	3.30	No change	24.5		
		Subsidence impacts	High	85.0	Moderate	10.0	Low	5.0		
	Low impact	EC_Change	High	24.1	Moderate	10.2	Low	55.7	Impossible	9.91
		Peat stability impacts	High	19.8	Moderate	17.2	Low	53.1	Impossible	9.91
		Inundation	Decrease	23.4	Increase	5.23	No change	71.3		
		Subsidence impacts	High	10.0	Moderate	20.0	Low	70.0		

A3 Conclusions

For each conceptual model type, BBNs were developed to model the sensitivity of the ecology of the:

- community as a whole, assuming generic impact upon the interactions of each of the species
- giant burrowing frog, recognising its specific habitat requirements in relation to changes in fire and flow regimes, even where peat impacts remain stable
- Blue Mountains water skink, recognising its specific habitat requirements in relation to decreases in inundation and water quality impacts
- giant dragonfly, recognising its specific habitat requirements in relation to changes to peat stability and watertable stability, which affect the lifecycle of the species
- spreading rope rush, recognising its specific habitat requirements in relation to nutrient enrichment as a result of decreased inundation and decreased water quality, which increases competition by invasive species.

The community/species modelled were most sensitive to peat stability impacts, inundation and fire. Inundation also has a strong influence on peat stability and fire, so is overall the strongest influence on sensitivity for most species BBN models. The BBN models for spreading rope rush differed from this result, and were more sensitive to vegetation, then inundation and fire and peat stability.

There was little difference in the relative sensitivities between the two types of BBN models for the headwater swamps and the hanging/valley infill swamps, with the fauna being most sensitive to peat stability, inundation and fire in both models. The difference between the two models was the impact of change in groundwater connection (and the resulting influence on inundation), since headwater swamps are unlikely to be connected to groundwater and so are not impacted by altered groundwater flow paths.

The scenario analyses showed the peat swamp ecological community and the individual modelled species were all impacted by the mining effects and associated ecosystem changes. The giant dragonfly appears the worst affected at high impacts but is also substantially affected with low impact. The water skink and the burrowing frog are also greatly affected when impacts are high but these levels can be reduced if the impacts are lowered. The peat swamp community had a similar effect. The rope rush is not as highly impacted as the other species.

Information linking subsidence effects to ecological impacts is limited, with little information that specifically describes how ecology responds to changes in the surrounding environment. In addition to this, there was no empirical evidence to assign probabilities to any of the relationships within the BBN. The BBN is therefore based on conceptualisation by specialists rather than on any measurement of impacts. This means the BBNs should be used as a risk assessment tool, rather than a definitive measurement of impact. It is important to recognise this limitation of the model, and to use the BBN results to:

- flag the risk of potential impact to community/species
- indicate areas for priority investigation.

The primary use of the model should be to design appropriate investigations to confirm the sensitivities suggested by the BBN, and to inform monitoring approaches in areas likely to be undermined. The BBN provides a framework that can be updated in the future as empirical evidence of impacts to peat swamps becomes available.

The importance of changes in inundation in determining impacts to peat swamps ecology indicates that management of the swamps should focus on maintaining natural levels and variations in inundation. Changes to inundation are most strongly influenced by subsidence impacts (via changes in groundwater connection for hanging/valley infill swamps). Therefore, to identify the potential for impacts to peat swamps early, monitoring needs to identify when subsidence impacts are occurring. That is, when cracking, tilting or fracturing of the sandstone substrate begins to occur, impacts to the peat swamps are imminent because subsidence usually results in reduced groundwater discharge to the swamp and/or leakage of water from the base of the swamp. The most effective trigger levels for monitoring will therefore apply to the level of fracturing within the sandstone. The strong influence of the ‘proximity’ node also indicates that the most effective tactic for reducing impacts is to locate mines distant from swamps.

The information on impacts, peat swamp function and sensitivity to altered physical environment will be used in the next project task to evaluate monitoring and reporting regimes, and to recommend a monitoring and reporting approach that can be employed by mine proponents to identify potential impacts before they occur.

It is important to remember that a time lag can occur between the time of subsidence impact to the detectable response exhibited by the community’s vegetation.

Table A4 Definitions for BBN nodes and states.

Nodes	Description	States	Description
Change in the ecology	Identifies changes to the ecology of the THPSS as a result of longwall mining. For the model of the ecological community as a whole, this node encompasses impact-induced changes to both the community's vegetation and biodiversity that are characteristic of the THPSS endangered ecological community. For the species models, this node identifies any change to the species, such as reduction in numbers, reduction in extent of coverage or declining population health. Although these systems are dynamic and can recover naturally after fires or erosion, this node represents the level of noticeable and long-term impacts that may occur due to longwall mining.	High degree of change	Significant changes to vegetation and/or biodiversity components of the swamp, such as significant shift in vegetation assemblage composition, significant change in terrestrial or aquatic habitat availability, or significant change in faunal assemblage. These changes will have a significant impact on the natural functioning of the community over a given time.
Note: For each mode, this node changes to represent change to the ecological community, giant burrowing frog, Blue Mountains water skink, giant dragonfly and spreading rope rush		Moderate degree of change	Some changes to the vegetation and/or biodiversity components of the swamp, such as shift in a component of the vegetation assemblage composition, change in a component of the terrestrial or aquatic habitat availability, or significant change in faunal assemblage. These changes will have an effect on the natural functioning of the community over a given time.
		Low degree of change	No discernible change to vegetation and/or biodiversity components of the community. These changes will have no noticeable effect on the functioning of the community over a given time.
		Impossible	Situations where the states for fire and inundation are not possible (i.e. where the states are not fire = decreased and inundation = increased, or vice versa).
Vegetation type	Refers to the natural vegetation types within the THPSS community, and uses vegetation type as an indicator of potential for impacts or resistance to impact. Vegetation types considered in this node are: • lower growing sphagnum bogs and fen vegetation types with shallower root networks and higher dependency on	Susceptible	A high proportion of smaller-growing vegetation types with shallower root networks and higher dependency on substrate moisture (e.g. sphagnum/sedge/rush/semi-aquatic grass dominated). These are the most common swamp vegetation types where conditions are wet for long periods.

Nodes	Description	States	Description
	substrate moisture • medium-growing sedge/herb/shrub-dominated vegetation types with moderately deep root networks and lesser dependency on substrate moisture • higher growing rush/shrub/tree-dominated vegetation types with relatively deep root networks and even lesser dependency on substrate moisture. Although ‘susceptible’ refers to the long-term potential for vegetation to demonstrate an impact, there is a time lag in vegetation response to change, and this varies for different species. Some species may ‘hang on’ for a long time (e.g. *Banksia robur*) and may therefore be considered unsusceptible. Other species may disappear quickly and therefore be considered susceptible. Susceptibility is strongly related to time lag. In this definition, there is little knowledge on specific species requirements that would allow susceptibility to be distinguished from time lag.	Moderate	A high proportion of medium-growing vegetation types with moderately deep root networks and lesser dependency on substrate moisture (e.g. sedge/herb/shrub dominated).
		Not discernibly susceptible	A high proportion of higher growing vegetation types with relatively deep root networks and even lesser dependency on substrate moisture (e.g. shrub/tree dominated). This vegetation can survive longer periods of dryness.

Nodes	Description	States	Description
Change in groundwater connection	Refers to the volume of discharge from the watertable aquifer to the swamp. This may be either from a perched aquifer or the regional aquifer. The volume of exchange depends on the height of the watertable relative to the swamp. Where increased permeability lowers the watertable, groundwater discharge to the swamp will also decrease. An increase in groundwater discharge is unlikely, but, if it occurs, it may also have deleterious effects on the swamp, since the swamp vegetation has evolved to a specific inundation regime and may be sensitive to any changes. This does not account for losses from the swamp to the watertable aquifer—those impacts effect the swamp inundation directly through subsidence impacts. This definition also includes recently infiltrated water that flows downwards through fractures and along lower permeability layers (sometimes called interflow), to discharge after a relatively short time. Widespread fracturing near the surface may cause this water to flow directly downwards rather than laterally, and so prevent the water from discharging to swamps on steep valley sides or cliff faces (i.e. hanging swamps).	Decreased discharge to swamp	Where groundwater discharge to the swamps decreases. This may be where a swamp goes from connected (gaining) to losing or disconnected. These scenarios occur due to a significant decline in the watertable. The ultimate result is that groundwater discharge to the swamp either decreases significantly, or ceases altogether. A significant decrease in discharge can occur even if the change in watertable level is relatively small. Decreased discharge is the most likely outcome where permeability has increased due to subsidence. Also refers to the change in flow path that can result from widespread fracturing near the surface, and which prevents recently infiltrated water from flowing along low-permeability layers in the sandstone and discharging at steep valley sides or along cliff faces (i.e. at hanging swamps).
		Increased discharge to swamp	Where groundwater discharge to the swamps increases. This may be where groundwater discharge into a connected gaining swamp increases. This scenario is considered unlikely to occur where subsidence has impacted permeability. It may also present a risk to the peat swamps, since increased groundwater discharge may alter the level on inundation, or the water quality in the swamps and some swamp species may be intolerant of such changes.
		No change	Where there is no change to the existing groundwater connection regime between the watertable and the swamp. This may occur if the swamp is disconnected from the watertable, so that there is no interaction to begin with, or where watertables remain stable so that discharges to the swamp or losses from the swamp do not change. It may also occur where fracturing in the rock near the surface does not alter flow paths for recently infiltrated water, such that the water continues to flow downwards and laterally, and

Nodes	Description	States	Description
			discharges to swamps on steep valley sides or cliffs.
Permeability	Refers to the permeability of the sandstone aquifers that underlie the swamps. Relates only to changes in permeability at the surface of the sandstone, where the sandstone is in contact with the base of the peat swamp. Increased permeability can occur through cracking caused by subsidence, and can cause the swamps to drain through the cracks in the sandstone.	Increased	Permeability increases at the surface of the sandstone, through cracking that propagates to the sandstone surface. These cracks remain open and act as a conduit for flow. Increased permeability is the most likely outcome when subsidence impacts are high.
		Same	No change in permeability at the surface. This means that either cracking does not occur, cracks do not extend up to the surface (so they are not in contact with the base of the swamp) or that cracks are filled with clay to retard water flow (although there is no empirical evidence that self-remediation through filling of cracks can occur).
		Decreased	Permeability at the surface of the sandstone is decreased. This outcome is very unlikely but could possibly occur through compaction of poorly consolidated material, or through filling of fractures with less permeable material than the original sandstone.
Subsidence impacts	The degree subsidence impacts at the land surface as a result of longwall mining. Refers to all types of subsidence impacts, including fracturing and cracking of sandstone at the surface (beneath the swamps); tilting of sandstone at the surface (beneath the swamps); valley closure, which includes collapse of cliffs and upsidence in valley floors (manifesting as cracking beneath swamps).	High	Significant cracking at the surface of the sandstone underlying the swamps. Significant tilting of the sandstone surface. Collapse of cliffs or significant fracturing within cliffs (most relevant for hanging swamps).
		Moderate	Some cracking at the surface of the sandstone, but swamp hydrology is not completely altered. Some tilting of the sandstone at the surface, but flow regime is only slightly altered. Cliffs are destabilised but do not collapse, and fracturing is minor and does not completely alter flow paths.

Nodes	Description	States	Description
		Low	Either there is no cracking that directly intersects the surface of the sandstone (and the base of the swamp) or, if the cracking does reach the surface, cracks are minor, closed or filled with sediment so that they do not provide a pathway for water flow. Tilting is minor enough that flow regime through the swamps is not altered. Collapse of cliffs does not occur, and any fracturing within cliffs is minor (closed fractures or quickly filled with low-permeability sediment) so that they do not provide a pathway for flow through the sandstone.
Geological characteristics	Incorporates all aspects of geology that make the substrate between the swamp and the mine more or less likely to be impacted by subsidence. That is, the geological characteristics that mean a rock type has a greater or lesser propensity for failure. This includes the lithology (rock type and degree of cementation, lithification and consolidation), presence of structural features (degree of fracturing and folding, bedding planes other discontinuities), presence of intrusions and mineralogy (occurrence of minerals known to impact water quality, such as siderite and marcasite). Where all these components are high, the geology is considered susceptible to subsidence impacts.	Susceptible	Geology between the mine and the swamp is brittle, with a high degree of insipient fracturing, jointing and folding; distinct bedding planes; presence of intrusions such as dykes; and containing a high proportion of iron-bearing minerals such as siderite and marcasite. The Hawkesbury Sandstone and Banks Wall Sandstone will largely fall into this state. Susceptible geology is likely to propagate the effects of subsidence upwards through the geological profile to the surface.
		Moderate	Geology is variable—some parts of the geological profile may be brittle while other layers may be more plastic. The occurrence of structural features such as insipient discontinuities (fractures, distinct bedding planes) or intrusion (such as dykes) is variable (present in some parts but not in others), or may be limited in extent and connectivity. Some iron-bearing minerals are present (such as marcasite and siderite).
		Unsusceptible	Geology is less brittle and more plastic; there are few insipient fractures, joints and folds; the rock structure is massive rather than having distinct bedding planes; dykes are not present; and iron-bearing mineral content is low (e.g. siderite and marcasite). Unsusceptible geology is less likely to propagate the effects of subsidence upwards through the geological

Nodes	Description	States	Description
			profile.
Mining depth	The depth of the longwall mining panel below the surface. This is effectively the vertical distance between the swamp and the longwall panels.	Shallow	The vertical distance between the swamp and the underlying longwall panel is minimal, and is therefore more likely to result in subsidence effects propagating to the surface. However, the angle of draw for a shallow mine means a smaller area at the surface would be impacted by subsidence. A shallow longwall mine in the Sydney Basin would typically be less than 300 m.
		Moderate	The vertical distance between the swamp and the underlying longwall panel is moderate, and therefore may result in subsidence effects propagating to the surface. However, the angle of draw for a mine at moderate depth means a moderately large area at the surface would be impacted by subsidence. A moderate depth longwall mine in the Sydney Basin would typically be between 300 and 500 m.
		Deep	The vertical distance between the swamp and the underlying longwall panel is significant, and is therefore less likely to result in subsidence effects propagating to the surface. However, the angle of draw for a deep mine means a larger area at the surface would be impacted by subsidence. A deep longwall mine in the Sydney Basin would typically be more than 500 m.
Channel incision	The surface topography where the swamp is located. Subsidence is likely to be worse in more steeply incised terrains than in flatter terrains. Therefore, hanging swamps and valley infill swamps are at higher risk of subsidence impacts than headwater swamps,	Incised	The swamp is located in incised terrain, such as on cliffs, steep valley sides or in incised creek beds. This type of topography is at higher risk of subsidence impacts.
		Moderate	The swamp is located in moderately incised terrain such as valley sides or moderately incised creek beds. Some valley

Nodes	Description	States	Description
	which occur in flat topography.		infill swamps fall into this category.
		Flat	The swamp is located in flat topography, such as the flat terrain near catchment divides. Headwater swamps largely fall into this category.
Dimensions	Dimensions of the longwall mine footprint, which includes consideration of panel length, width and height; number of panels; and width of pillars between the panels). Aims to characterise the proportion of material removed from the mine.	Large footprint	Longwall panels are wide, long and high, with small pillars remaining between each panel. The proportion of material removed is high compared to the overall footprint of the mine.
		Moderate footprint	Longwall panel dimensions are variable or moderate. The proportion of material removed is moderate compared with the material remaining in the pillars.
		Small footprint	Longwall panels are narrow, short and thin, with larger pillars remaining between each panel. The proportion of material removed is low compared with the overall footprint of the mine.
Proximity	Indicates the lateral distance between the mine and the swamps at the surface, as swamps that are further from the underlying longwall panels are generally at lower risk of impacts than those directly above the longwalls. Also incorporates an understanding of where the swamps are with respect to the mine footprint.	Edge	Swamps occurring at the edge of longwall panels. Differential strains occur at the edge of longwall panels, which cause the land surface to subside unevenly. This results in greater cracking, fracturing and displacement at the surface, and is therefore more likely to cause impacts on swamps. This state defines the 'edge' as the point vertically above the edge of the longwall panels radiating out to a distance of 1 km.
		Over	Swamps occurring within the footprint of the longwall panels. The strain is more evenly distributed directly above the longwall panels and subsidence is therefore more uniform at the surface. That is, the land surface subsides evenly and may not cause significant cracking or fracturing at the surface. Swamps directly above the mines may therefore be partially protected from subsidence impacts.

Nodes	Description	States	Description
		Distant	Subsidence impacts diminish with distance from the edge of the longwall panel. Therefore, swamps that are distant from the longwall mine are less likely to be impacted by subsidence. Distant is considered to be greater than approximately 1 km from the edge of the longwall mine.
Peat stability impacts	The stability of the peat within the swamp, in relation to the potential that the peat substrate will erode due to changes in flow regime, fire intensity and frequency, and levels of inundation.	High	Peat is drier than usual and cracks, fissures and nick points have developed within the peat. Erosion is occurring along channels (channel width is increasing) and at the downstream edge of the swamp. Peat may have been transported downstream as peat balls.
		Moderate	Some cracking and fissuring of the peat is evident due to slightly drier peat conditions. No new nick points are evident and erosion has not occurred.
		Low	Peat is stable, with no new cracks, fissures or nick points. Channels remain narrow relative to their depth. No evidence of peat erosion.
Peat composition	The proportion of peat in the swamp compared with the proportion of inorganic material such as sand, silt, clay or gravel. The organic content influences the intensity of fires that burn through the swamps, because a peat with higher organic content is more likely to catch alight and continue to burn.	High organic matter	Organic content higher than 50% of total swamp substrate.
		Low organic matter	Organic content of less than 50% of total swamp substrate. Most swamps are likely to fall in this category.
Fire risk	The intensity and frequency of fires that impact the swamps. Fire risk is directly dependent on the level of inundation. A decrease in swamp inundation increases the risk of more intense fires that burn the swamp vegetation and the peat. It also means that fires will burn through the swamps more frequently, as under normal	Increased risk	The likelihood of more frequent and intense fires burning through the swamp increases, and both vegetation and peat are likely to burn. It also means that swamps passing through the surrounding countryside are more likely to burn through the swamp. This occurs because swamp inundation has decreased, making the swamp more prone to fires. This is the most likely scenario in response to subsidence impacts.

Nodes	Description	States	Description
	levels of inundation the moisture in the swamps largely protect them from fires burning through the surrounding country. An increase in swamp inundation can decrease the fire risk, since the wetness of the swamp means fires are less likely to burn through it or, if they do, are likely to be less intense. Any other combinations of fire risk and inundation are impossible.	No change	The current level of fire risk to the swamp is maintained. The moisture of the swamps largely protects them from fires. Fires that do burn through the swamps are low intensity and may only burn the top of the vegetation. The frequency of fires burning through the swamps does not change.
		Decreased risk	The risk of fires burning through the swamps decreases, due to an increase in inundation. Fires that burn through the swamp are less frequent and less intense than usual, as the swamp is protected by its wetness. Fires do generally not burn the swamp vegetation at all, but may damage vegetation at the perimeter of the swamps. This scenario is considered very rare.
Flow regime	Flow regime characterises how water flows through the swamp, and includes consideration of both the flow paths through the swamps and the velocity of flow through the swamps. For example, flow paths may be infiltration through the peat, or through in-channels on either side of the swamp or through the centre of the swamp. Changes to flow paths can destabilise the peat (through development of nick points and subsequent erosion). Increased velocity of flow through the swamps can also cause erosion. Changes to the flow regime can occur due to subsidence impacts (tilting). Flow regime is separate from any consideration of the volume of water flowing through the swamp.	Major change	Significant changes to swamp flow regime. This may manifest as flow paths/channels changing from one location to another within the swamp and may result in the development of nick points. It could also be a change in the morphology of channel flow in the swamps (e.g. from sheet flow across the swamp to channels, or from meandering channels to straighter channels). The gradient of the swamp may also change, resulting in an increased slope and higher velocity flow through the swamp. This may also cause development of nick points in the swamp. A high level of change to flow regime is likely to incorporate most of these impacts.
		Minor change	Some changes to the swamp flow regime, potentially including altered flow paths/channelling through the swamp where channels change location or morphology (e.g. from sheet flow across the swamp to channels, or from meandering channels to straighter channels). It may also be a slight increase in gradient, which results in slightly increased flow velocity. A moderate change to flow regime incorporates some of these changes, but not all.

Nodes	Description	States	Description
		No change	No discernible changes in flow paths through the swamp, no development of new channels and no change of channel location within the swamp. No discernible increase in velocity of flow through the swamps.
Inundation	Water level in the swamp with respect to the swamp surface. Changes in inundation have a strong influence on swamp stability, vulnerability to fire and vegetation composition. The level of inundation is likely to vary seasonally in response to rainfall; however, this node refers to changes from normal conditions. Incorporates consideration of changes to the duration, frequency and interval of inundation, since these may all have negative impacts on the swamp.	Decreased	The water level in the swamp declines compared with normal levels so that the water level is beneath the peat surface either permanently or for longer than usual. This can occur due to changes in surface inflow, changes in groundwater discharge to the swamps or subsidence impacts (which either change the surface inflow to the swamps or drain water from the base of the swamps through cracks). Decreased inundation makes the swamp more prone to intense fires, peat instability (erosion) and altered vegetation composition. Peat will also desiccate, oxidise and lose some of its water-absorbing capacity if inundation decreases.
		Increased	The water level in the swamp increases compared with normal levels so that the water level is above the peat surface either permanently or for longer than usual. This can occur due to changes in groundwater discharge to swamps. The link between subsidence impacts and increased water levels is uncertain, but is likely to have a low probability of occurring. Increased inundation makes the swamp more resistant to fires but can alter vegetation composition and peat stability.
		No change	The water level in the swamp does not change from the normal patterns of inundation.
Surface inflow	Change in the volume of water contributed to the swamp through surface run-off, including direct rainfall and run-off from surrounding catchment (overland flow and up-catchment streams). Assumes that both an increase and a decrease in surface run-off will have	Decrease	The surface run-off contribution to swamp hydrology is less than the long-term average. In this case, subsidence impacts have decreased the volume of inflow reaching the swamps, and the normal surface inflow volumes (determined by catchment size and rainfall) have been insufficient to buffer against the changes caused by subsidence.

Nodes	Description	States	Description
	negative impacts on the swamp. Rainfall and catchment size determine the volume of the surface inflows and hence the interpreted buffering against changes to swamp hydrology. Subsidence impacts can influence changes in surface inflow where cracks occur in the catchment above the swamp. Since the swamps are high in the catchment, the probability of this occurring is relatively low.	Increase	The surface run-off contribution to swamp hydrology is greater than the long-term average. In this case, subsidence impacts have increased the volume of inflow reaching the swamps. The probability of this occurring is minimal.
		No change	The surface run-off contributions to swamp hydrology are consistent with long-term averages. This is the most likely scenario, since the probability that subsidence impacts intercept surface flow in the catchment above the swamp is low (as most swamps are high in the catchment).
Catchment size	Total upstream area that contributes run-off to the peat swamps. Catchment size partially determines the volume of surface run-off contributed to the swamp, and therefore its level of inundation.	Small	Small upstream catchment area, meaning the volume of run-off contributed to swamps is also likely to be small.
		Medium	Medium-sized upstream catchment area, meaning the volume of run-off contributed to swamps is likely to be moderate.
		Large	Large upstream catchment area, meaning the volume of run-off contributed to swamps is likely to be large.
Rainfall	Rainfall compared with the range in rainfall across the whole swamp community. Rainfall partially determines the volume of recharge to the swamp and therefore its level of inundation. Swamps in higher-rainfall areas may be better able to cope with other changes to swamp hydrology, because the large rainfall volumes buffer against impacts. Similarly, in lower-rainfall areas, swamps may be more sensitive to changes in hydrology. The volume of rainfall is combined with catchment size to	Low	Rainfall is low compared with the average rainfall received by swamps in this community. The swamps have limited ability to cope with other changes in hydrology, because the lower rainfall provides little buffering capacity. This limited buffering capacity is more acute during drought, when impacts to the swamps will be exacerbated.
		Moderate	Rainfall is moderate compared with other swamps in the community—that is, some swamps receive higher rainfall and others receive lower rainfall. These swamps have a moderate capacity to cope with other changes in hydrology, because some buffering is provided by rainfall.

Nodes	Description	States	Description
	determine the relative surface inflow to swamps.	High	Rainfall is high compared with the average rainfall received by swamps in this community. The swamps have a better ability to cope with other changes in hydrology, because significant buffering capacity is provided by the high rainfall. This buffering capacity is removed during drought, which is when impacts to the swamp are more acute.
Water quality impacts	The changes in water quality as a result of subsidence, such as increased iron content, iron staining, mats and methane bubbles. These changes can occur when cracking caused by subsidence exposes new rock surfaces to water flow, resulting in dissolution of iron and manganese-rich minerals. Note that this applies only to water quality within the swamps and not water quality in downstream waterways. No evidence of water quality changes in the swamps has been reported in the literature (but many impacts have been reported for the downstream waterways). Water quality changes in peat swamps are considered to be relatively unlikely, because of the small amount of flow from higher in the catchment and the function of peat swamps as a water filter for the downstream catchment. Impacts to water quality resulting from discharge of mine waste water has not been considered in the BBNs, since discharge does not occur above the swamps.	High	Significant impacts on water quality within the swamp, including iron matting and methane bubbles.
		Moderate	Some impacts on water quality, such as some iron and manganese precipitation, minimal formation of iron mats, and some methane bubbling.
		Low	No discernible changes to swamp water quality.
Water quality susceptibility	Refers to the natural chemical composition of water flowing into the swamp, which controls the propensity for water quality impacts. For example, the natural water quality may mean it	High	Natural water quality flowing into the swamps is likely to be impacted from chemistry changes as a result of subsidence impacts. The water does not have inherent buffering capacity against impacts such as iron matting or discolouration.

Nodes	Description	States	Description
	is not susceptible to further impacts as a result of subsidence.	Low	Natural water quality flowing into the swamps is unlikely to be impacted from chemistry changes as a result of subsidence impacts. The water may have inherent buffering capacity against impacts such as iron matting or discolouration.

Appendix B: Types of upland swamps in the Sydney Basin—Dr Ann Young

This table has been provided by peer reviewer Dr Ann Young to demonstrate the different terminology used to describe upland swamps in the Sydney Basin. Swamp types have been grouped into four main categories, however within each category there are a range of different terms and descriptions that have been used in the literature. Young's preferred typology and nomenclature is shown in bold.

Table B1 Different terminology used to describe upland swamps in the Sydney Basin.

Swamp type	Description	Source
Hanging swamp	Occur mainly in the Blue Mountains area, where the sandstones of the plateau alternate with extensive claystones and dissection by streams exposes both lithologies on the valley sides. They are fed by seepage through the sandstone (via the joints and bedding planes that break that sandstone into large blocks), which then emerges on the cliff face of the valley side when it reaches much less permeable underlying claystone. Hanging swamps occur on quite steep, sometimes near-vertical slopes. They have only shallow or minimal sediment and are essentially a thick mat of shrub and fern vegetation.	DSEWPaC (2012)
Headwater swamp - definition 1 (Also referred to as valleyside swamps, 'headwater-drainage divide swamps', and 'low slope headwater valley swamps')	The following definition is consistent for all three references listed in the right hand column, however three different swamp names as used: Occur on all plateaus in the Basin. They are found very close to the watersheds between the shallow headwater valleys on the plateau surfaces. Occasionally they overlap these ridges in areas of very low relief. They form when sandy sediment eroded from the plateau accumulates on the very gently sloping surface and on benches. They often have outcrops of sandstone within their boundaries, sometimes marked by groves of small eucalypts. Elsewhere the soils are shallow (less than 1 m deep), dark grey from accumulated organic matter and resting on coherent	Young 1986a (uses 'valleyside swamp') NSW Bulli Seam PAC 2010 (uses 'headwater swamp') Tomkins & Humphrey 2006 (uses 'headwater-drainage divide swamp') DSEWPaC (2012) (uses 'low slope headwater valley swamp')

Swamp type	Description	Source
	sandstone. They usually have a low shrub/sedge swamp vegetation. They may merge downslope with valley floor swamps.	
Headwater swamp - definition 2	*Headwater swamps are the significant majority of the upland swamps and are generally situated in areas near catchment divides where plateau incision is weak and topographic grades are shallow.* *These upland swamps can be quite extensive and 'drape' over the undulating Woronora Plateau. They can fill shallow valley floors and extend up the valley sides and drainage lines to straddle catchment divides in areas of shallow, impervious substrate formed by either the bedrock sandstone or clay horizons*.(NSW DP 2008) This definition includes many valley floor swamps and while it is attributed in the NSW DP 2008 to Young (1986a) it misinterprets that reference because Young separated valleyside (=headwater swamps definition 1) from valley floor swamps.	NSW DP 2008 a similar definition was used by Helensburgh Coal Pty Ltd 2008
Headwater swamps—definition 3	*Headwater swamps occur within broad scale, relatively low slope creek or tributary headwater areas. A significant body of evidence indicates that most, if not all headwater swamps are 'embedded' in a broader scale 'hillslope aquifer' which provides the excess of precipitation over evapotranspiration (ET) which sustains them i.e. they are predominantly groundwater fed over the long term (including through droughts)* (EcoEngineers 2012, p. ix). This definition differs from most researchers who see headwater swamps as rain-fed and disconnected from groundwater.	EcoEngineers 2012
Valley floor swamps (Also referred to as valley bottom swamps, valley filling swamps, valley fill swamps, valley in-fill swamps and 'in-valley'	The following definition is consistent for all five references listed in the right hand column, however four different swamp names as used: Valley floor swamps occur in the floors of shallow headwater valleys	Young 1986a (uses 'valley floor swamps') DSEWPaC 2012 (uses 'valley bottom swamps')

Swamp type	Description	Source
swamps)	on the plateau surface. Sandy sediment accumulates because the discharge of these small low-gradient streams is too low to shift the sand through the valley into the gorges below. Water accumulates in the sediment from rainfall and from seepage out of bedding planes in the adjacent valley sides, and perhaps also from shallow groundwater aquifers in the sandstone. The soils range in both organic matter content and texture from shallow dark grey sandy soils on the margins to deep (2 m or more) black fine-grained sediments in the valley axes. The vegetation can be diverse, often changing from low shrubs on the margins to sedgeland to tall shrub thickets along the valley axes.	Tomkins & Humphrey 2006 (uses 'valley filling swamps') NSW Bulli Seam PAC 2010 (uses 'valley fill swamps') Helensburgh Coal P/L 2008 uses 'in-valley' swamps
(this valley in-fill definition is equivalent to valley floor swamps in other studies, but not equivalent to valley infill swamps as used in other studies)	*Valley in-fill swamps are those which fringe, and have arisen from, sand accumulation along well defined streams where there is a potential for scour of the sandy substrate of the swamp(s) above a certain stream power and erosive resistance threshold. The changes in grade that may result from mine subsidence are only likely to induce excessive shear in relatively low gradient swamps. Therefore the swamps at risk from scour and erosion as a result of longwall mining are those where the stream is of a high order (i.e. high flow and low gradient), has poor vegetation condition (e.g. from drying and/or bushfire damage), and the longwalls lie perpendicular to the long axis of the swamp (Ecoengineers 2012, pp viii-ix).*	Ecoengineers 2012 (uses 'valley in-fill swamps')

Swamp type	Description	Source
Valley infill swamps (Also referred to as 'isolated pockets of valley-filling swamps')	The following definition is consistent for both references listed in the right hand column, however two different swamp names as used: Like valley floor swamps but occur downstream of, and separated from, other swamps. They probably form when a valley floor swamp is eroded, a slug of sand is transported downstream but not as far as the gorges, and a swamp is established on the deposited sand. *These 'valley infill' swamps form as isolated pockets blanketing the floor of incised second or third stream valleys and therefore tend to be elongate downstream (Tomkins and Humphreys 2006). They are believed to be initiated by rapid transportation of sediment material downstream and equally rapid deposition possibly as a result of channel profile-restriction (e.g. by log jams). Once initiated, the swamps are probably self-reinforcing, trapping more sediment, raising the water table and fostering the growth of organics and formation of peat (Tomkins and Humphreys 2006). Examples include Flatrock Swamp, on Waratah Rivulet above Metropolitan Colliery, Swamps 18 and 19 on Native Dog Creek above Elouera Colliery and Martins Swamp above the closed Nebo Colliery* (NSW DP 2008, p. 16). (NB. Ann Young's opinion is Swamps 18 and 19 and Martins Swamp are valley floor NOT valley infill swamps)	NSW DP 2008 (uses 'valley infill swamps') and Ann Young has adopted this distinction from valley floor swamps, originally proposed by Tomkins and Humphrey 2006 Tomkins & Humphrey 2006 (uses 'isolated pockets of valley filling swamps') Note: Helensburgh Coal P/L 2008 includes these in 'in-valley' upland swamps

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